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Habitat use and species assemblage of bats in a northeastern coastal plain ecosystem

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HABITAT USE AND SPECIES ASSEMBLAGE OF BATS IN A NORTHEASTERN COASTAL PLAIN ECOSYSTEM

by

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ABSTRACT

J.J. Fletcher. Habitat Use and Species Assemblage of Bats in a Northeastern Coastal Plain Ecosystem, 107 pages, 12 tables, 5 figures, 3 appendices, 2017.

Bat habitat use in the Northeast has been well studied, however, research has been focused on interior locations, leaving coastal areas relatively understudied. Cape Cod National Seashore (CCNS) is a coastal plain peninsula where our understanding of bat habitat associations is limited to historical data. I acoustically sampled sites within CCNS during 2015 and 2016 to quantify local and landscape factors associated with habitat use and species assemblage. I examined interspecific effects of co-occurrence between two bat species to understand factors influencing habitat use in CCNS. I found that coastal bats use sites similar to interior populations despite differences in dominant vegetation types. *Myotis septentrionalis* were found in relatively high numbers and availability of suitable habitat or competition with other species were not limiting factors to recovery on CCNS. Continued acoustic monitoring on CCNS should extend beyond summer habitat use and coordinate mist netting to improve power to draw inferences.

Keywords: Acoustic monitoring, Cape Cod National Seashore, coastal plain, co-occurrence, habitat use, *Myotis septentrionalis*, species assemblage

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CHAPTER 1: HABITAT USE BY LOCAL BAT SPECIES IN THE NORTHEASTERN ATLANTIC COASTAL PLAIN ECOSYSTEM

THREATS TO NORTHEASTERN BAT SPECIES

Recently, much attention has focused on conservation and management of bats due to ecosystem services they provide and for their role as indicator species (Fenton 1997, Kunz et al. 2011). Current bat populations are threatened by exposure to pesticides (Clark 1988), renewable wind farms (Baerwald and Barclay 2011), destruction of foraging areas and summer roosting habitat (Henderson et al. 2008), and disturbance to winter hibernacula (Johnson et al. 1998). In recent years, bats in the Northeast and Midwest United States have been decimated by white-nose syndrome, a disease caused by the psychrophilic (i.e. cold-loving) fungus *Pseudogymnoascus destructans*. White-nose syndrome has killed millions of bats since its discovery in New York in 2007 (Frick et al 2010). As of March 2017, presence of *P. destructans* has been confirmed in 33 states. The distribution of *P. destructans* ranges from Maine throughout the Northeast and Midwest, as far west as Texas and as far south as Georgia (White-nose Syndrome 2017). Infected bats exhibit unusual hibernation behaviors including frequent and long periods of arousal from torpor, and premature exit of hibernaculum caves to forage (Warnecke et al. 2012). Small insectivorous bats that use caves to hibernate in large colonies are most affected by this disease. The proximity of individuals in these colonies result in a high rate of bat-to-bat transmission. Furthermore, cold, moist cave environments are ideal for growth of *P. destructans*. The northern long-eared bat (*Myotis septentrionalis*) is a small, insectivorous bat historically common in the eastern United States and Canada that hibernates in cave environments in large numbers. Decline in this species due to white-nose syndrome led to a recent decision by the U.S. Fish & Wildlife Service to list *M. septentrionalis* as threatened under

the Endangered Species Act. Since then, listing has been proposed for two other declining species, little brown bats (*Myotis lucifugus*) and tricolored bats (*Perimyotis subflavus*). Since the onset of white-nose syndrome, recent attempts to improve understanding of species distribution and habitat use have been limited by low population numbers.

HABITAT USE AND SPECIES CO-OCCURRENCE OF BATS IN THE NORTHEASTERN UNITED STATES

Habitat requirements for Northeastern are reliant on distinct habitat requirements that differ seasonally. Recent studies have prioritized behavior and mortality during winter hibernation instead of summer activity and habitat use due to the psychrophilic tendencies of *P. destructans*. Post-white-nose-syndrome, research has become limited due to low population numbers. A better understanding of habitat requirements is needed to facilitate population recovery that will require high fecundity and recruitment in these species.

In summer months, bats choose resources and conditions to fulfill both foraging and diurnal roosting requirements. Summer bat ecology within unique ecosystems of the Northeast, such as the Atlantic coastal plain, both pre- and post-white-nose-syndrome is poorly understood. Information on bat ecology in the northeastern Atlantic coastal plain is limited to historical data collected in the early 1900's (Murphy and Nichols 1913, Nichols and Nichols 1934). Conversely, southeastern areas of the Atlantic coastal plain have been well studied. Research conducted in the southeastern Atlantic coastal plain suggests that bat species in the genus *Myotis* are unlikely to use coastal plain ecosystems for foraging and roosting (Furlonger et al. 1987, Menzel et al. 2000). However, a 2012 survey on Long Island, New York (within the northeastern Atlantic coastal plain ecosystem), captured 15 *M. septentrionalis*. This was the largest number of *M.*

septentrionalis captured in all Northeastern states that year, indicating that post-white-nose-syndrome *Myotis* populations may use coastal plain habitats in the northeast (Fishman unpublished data). We know little about bat assemblages and habitat ecology in the northeastern Atlantic coastal plain ecosystem, yet this information is necessary for successful bat management and conservation in this unique region.

There are nine bat species known to historically occur within the northeastern United States. Of these nine, five species are state listed in Massachusetts as endangered: northern long-eared bats (*Myotis septentrionalis*), little brown bats (*Myotis lucifugus*), eastern small-footed bats (*Myotis leibii*), Indiana bats (*Myotis sodalis*), and tricolored bats (*Perimyotis subflavus*). Of these state listed species, *M. sodalis* and *M. septentrionalis* are also federally listed under the Endangered Species Act. The other four species that historically occur in Massachusetts are big brown bats (*Eptesicus fuscus*), Eastern red bats (*Lasiurus borealis*), hoary bats (*Lasiurus cinereus*), and silver-haired bats (*Lasionycteris noctivagans*).

Myotis Species

In the inland northeast, there are four small-bodied, insectivorous *Myotis* bat species: *M. septentrionalis*, *M. lucifugus*, *M. sodalis*, and *M. leibii*. *M. leibii* are among the rarest bat species in North America and generally occur in mountainous regions (Best and Jennings 1997). They use caves and rock outcrops for winter hibernation and summer diurnal roosts. Neither of these features are prominent on the Atlantic coastal plain landscape. *M. septentrionalis* are relatively small bats with low wing loading (mass/area of wing) and low wing aspect ratio (length of wingspan squared divided by total area of the wings; Farney and Fleharty 1969). These characteristics allow *M. septentrionalis* to forage in forests with high levels of vertical structure

(i.e. clutter) where they specialize in gleaning insects from forest substrate (Caceres and Barclay 2000). *M. septentrionalis* prefer to use trees as diurnal roosts during summer months in the northeastern United States, but may use other structures such as houses and telephone poles (Caceres and Barclay 2000, Carroll et al. 2002). In comparison, *M. sodalis* are federally listed as endangered, small-bodied bats with low wing loading and low wing aspect ratio allowing for high maneuverability (Farney and Fleharty 1969). *M. sodalis* also occupy a specialized niche that includes diurnal tree roosting and interior forest foraging (Thomson 1982, Menzel et al. 2005). There is a paucity of research addressing habitat use of these forest-roosting bats in the Northeast (Fenton et al. 1992, Miller et al. 2003).

In contrast to the specialized *M. septentrionalis* and *M. sodalis*, *M. lucifugus* are a generalist species that occupy a variety of summer diurnal roosts and forage in a variety of land cover types. *M. lucifugus* will commonly use urban structures for roosting during summer months, but have also been found in trees, under rocky substrate, or in caves (Fenton and Barclay 1980). Relatively low wing loading and low wing aspect ratio allow these bats to maneuver efficiently for aerial hawking of flying insect prey and gleaning from substrates (Ratcliffe and Dawson 2002). While not in the same genus, *P. subflavus* are a generalist species that have similar habitat requirements as *M. lucifugus* (Fujita and Kunz 1984).

All *Myotis* bats and *P. subflavus* hibernate in caves making them susceptible to the psychrophilic *P. destructans* and all five species have been identified with white-nose syndrome. *M. septentrionalis* and *M. lucifugus* have faced up to 90% declines in winter hibernacula. Populations of *P. subflavus*, *M. leibii*, and *M. sodalis* have also declined due to white-nose syndrome.

Eptesicus fuscus

Often described as a generalist species, populations of big brown bats (*Eptesicus fuscus*) in inland regions of the northeast have been known to use a variety of land cover types. While most bats avoid heavily developed areas, *E. fuscus* has been shown to use these areas advantageously by foraging in well-lit environments that attract insects and roosting in man-made structures during summer months (Furlonger et al. 1987, Kurta and Baker 1990). Large body size, high wing loading, and high wing aspect ratio reduce maneuverability which may be why *E. fuscus* choose to forage in more structurally open areas for ease of travel (Farney and Fleharty 1969). *E. fuscus* tend to hibernate individually or in small groups during winter months and are found in caves, as well as buildings or other man-made structures. While *E. fuscus* individuals with white-nose syndrome have been recorded, overall species populations appear less affected by the disease than colonially hibernating species.

Tree Bats

Summer habitat use of northeastern migratory bat species is poorly understood. *L. borealis, L. cinereus,* and *L. noctivagans* are all relatively large species that range from slightly to highly migratory, but are known to occur within Massachusetts at some point during the year (Cryan 2003). Unlike most northeastern bat species that form summer maternity colonies, all three species are thought to roost in solitary diurnal tree roosts (Kunz 1982, Shump and Shump 1982a, Shump and Shump 1982b). Due to large body size, high wing loading, and high wing aspect ratio, *L. borealis*, *L. cinereus,* and *L. noctivagans* maneuver poorly though structurally cluttered land cover types. *L. borealis* prefer more open and well-lit areas to forage, but avoid urban and heavily developed land cover (Furlonger et al. 1987, Walters et al. 2007). In contrast,

L. noctivagans habitat use, like *E. fuscus*, is often positively associated with developed or urban environments (Gehrt and Chelsvig 2004). Overall, these larger bats tend to use areas that facilitate ease of movement for successful insect foraging.

BATS WITHIN THE ATLANTIC COASTAL PLAIN ECOSYSTEM

The Atlantic coastal plain is a unique environment that runs from the southern tip of Florida to Cape Cod, Massachusetts at its northernmost boundary. This coastal plain ecosystem is characterized by flat, low elevation land adjacent to the Atlantic Ocean. A variety of disturbance regimes, from frequent fires in drier habitats to floods and hurricanes in wetter climates, define and develop the variety of land cover types in this system. In the northeastern United States, the Atlantic coastal plain supports an array of flora and fauna and serves as the range limit for many terrestrial species. Cape Cod National Seashore lies at the northern end of the Atlantic coastal plain and serves as a last protected area for species at their range boundary. For example, Eastern box turtles (*Terrapene carolina carolina*) are in decline throughout most of their range, but remain relatively common within Cape Cod National Seashore. Although recent surveys by National Park Service biologists at Cape Cod National Seashore suggest that the seashore contains some of the densest populations of Eastern box turtles in the state of Massachusetts, they continue to lack adequate information on species distribution and occurrence (Cook 2014). These informational barriers hinder management and protection for a species of concern that may behave differently at the edge of its range and in a coastal plain ecosystem. Despite its unique status, the full extent of floral and faunal habitat and support that the Atlantic coastal plain ecosystem provides is unknown. Bats are another group of species known to persist within the northeastern regions of the Atlantic coastal plan, including Cape Cod National

Seashore, but are understudied in terms of habitat use and distribution in this area. This study is one of a few comprehensive assessments of summer bat habitat use and the only research on species co-occurrence in a northeastern Atlantic coastal plain ecosystem.

The most recent research detailing bat species presence and habitat use within the northeastern Atlantic coastal plain region were conducted on Martha's Vineyard and Camp Edwards Military Training Installation, both of which are on the western boundary of Cape Cod and outside of the managed areas of Cape Cod National Seashore (Buresh 1999, Kelly and Ciaranca 2000). Miller (1897) recorded three species of migrating bats within the national seashore over one hundred years ago. There has been no published work within Cape Cod National Seashore on bat species presence or habitat use since this observational study. The National Park Service at Cape Cod National Seashore has multiple land use requirements and management techniques to maintain this popular tourist destination with a variety of wildlife and historical land cover types. Frequent prescribed burning, conducted to mimic historical disturbance events and maintain habitat for heathland and grassland species, is a common management practice in the national seashore. An annual average of four million visitors to the national seashore results in a high level of human disturbance and the necessity for potential human-wildlife conflict management. The National Park Service requires current species assemblage and habitat use information to avoid conflicts with current management practices.

RESEARCH GOALS

Habitat use modeling enables managers to understand current distributions for species of interest, predict future distributions, and focus management objectives (Morrison 1992). Modeling habitat use of bats in this ecosystem will update our understanding of species' habitat

associations and distributions in Cape Cod National Seashore which represents an understudied region and ecosystem. Investigating species co-occurrence will show how bat species are interacting in this ecosystem. In addition to the specific habitat resources and conditions that define diurnal roost and summer foraging needs for northeastern bat species, species interactions may play a role in shaping habitat use. Many studies in the United States have focused on local conditions associated with habitat use by insectivorous bats, but information on the role of competition and co-occurrence on habitat use is less developed. The goals of this study are to 1) identify all bat species present on Cape Cod National Seashore, 2) determine what local and landscape conditions are associated with habitat use for these bats, and 3) assess how use of an area by one species may be associated with use by another in this unique region. This information will be beneficial to natural resources managers at Cape Cod National Seashore and in other areas of the Northeast as they design management objectives for proactive habitat improvement for declining bat species living within coastal plain and developed ecosystems.

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CHAPTER 2: BAT SPECIES ASSEMBLAGE AND FEATURES ASSOCIATED WITH HABITAT USE IN CAPE COD NATIONAL SEASHORE

ABSTRACT

Assessment of habitat use at multiple spatial scales is necessary for effective management for species of concern. Cape Cod National Seashore (CCNS) is a large, federally protected area that hosts a variety of floral and faunal species of interest, but information of bat habitat use is over 100 years old. In 2015 and 2016, I acoustically monitored bat populations within CCNS and used remote sensing data and vegetation surveys to quantify local and landscape factors associated with habitat use and species assemblage. I used an occupancy framework to model bat species habitat use within Cape Cod National Seashore and compared *a priori* models using Akaike's Information Criterion adjusted for small sample size or overdispersion depending on goodness-of-fit (AICc or QAICc). Most bat species used landscape and local resources and conditions similar to what has been shown in interior populations of the northeast. The northern long-eared bat (*Myotis septentrionalis*) has become a species of concern due to the recent listing as federally threatened. *M. septentrionalis* habitat use was significantly negatively associated with the dominant pitch pine (*Pinus rigida*) vegetation type and sample year indicating a potential population decline for this species. The top model also showed that *M. septentrionalis* site use was positively associated with proximity to freshwater sources. To encourage *M. septentrionalis* persistence in this coastal ecosystem, CACO managers should focus management efforts on preserving freshwater kettle ponds and encouraging mixed forest vegetation types for *M. septentrionalis* foraging and roosting.

Keywords: Cape Cod National Seashore, coastal ecosystems. habitat use, *Myotis septentrionalis*, occupancy, species assemblage

INTRODUCTION

To effectively manage for species of interest, an understanding of habitat use is critical and provides insight into species distribution and factors influencing survival and reproduction (Morrison 1992). Habitats are the species-specific resources and conditions in any given area that allow an individual to occur, persist, and reproduce. Habitat use is measured by the occurrence and distribution of individuals within available potential habitat (Morris 1987, Krausman 1999). Habitat requirements may vary between or within seasons depending on the ecology of the species. For most temperate bat species, habitat use requirements in summer are driven by resource availability for successful foraging and diurnal roosts, whereas winter habitat use depends on conditions necessary for prolonged hibernation (Fenton 1990). Preparation for future effects of management plans necessitates landscape, local, and temporal descriptions of habitat use requirements for species of interest to anticipate and mediate conflicts with management goals (Morrison 1992).

Because individuals make habitat selection decisions at multiple scales, where landscape decisions shape local level choices, assessing habitat use within discrete spatial scales is essential for effective management and conservation of target species (Johnson 1980, Orians and Wittenberger 1991). At the landscape scale, many bat species use roosting and foraging sites close to water sources for both hydration and increased insect abundance for foraging (Barclay 1991). Generalist species like *Eptesicus fuscus* and *Myotis lucifugus* make selections based on availability of, or proximity to, roadways due to their use of man-made structures for roosting (Fenton and Barclay 1980) or improved foraging (Furlonger et al. 1987). At the local level, bats may select to use sites within their chosen landscape based on resources or conditions, such as tree species, canopy height, or amount of vertical structure (i.e. clutter). Based on species

ecology, these resources and conditions are selected because they enable successful foraging, roosting, or facilitate easy travelling to and from roosting and foraging areas. Components of bat morphology, such as body mass and wing loading, prey preferences, and other species interactions also play a role in determining which local conditions a bat species uses. For example, low wing loading and small body size allow bat species to maneuver and forage in conditions with a high amount of understory and mid-story vertical structure (Aldridge and Rautenbach 1987). Larger bodied bat species choose or are restricted to foraging in more open areas due to reduced maneuverability, but can consume a wider variety of larger prey options. These morphological components and species-specific preferences together shape local habitat use decisions and patterns. Although landscape-level selection determines overall distribution of a species, local-level habitat use decisions influence the placement of foraging areas, roosting areas, and home ranges within a landscape. And although some land management may be planned broadly at the landscape scale, most management actions are applied on smaller scales.

Investigation of species occupancy has become a useful tool for resource managers working to assess population status of species of interest based on distribution, probability of occupancy, and features associated with occupancy. Occupancy analysis determines the probability that a species occupies a specific location and the probability of detecting that species given that the site is occupied (MacKenzie et al. 2002). Although occupancy analysis does not include estimates of abundance, estimates of occupancy are often positively associated with abundance and are a viable alternative when abundance assessment is not feasible (MacKenzie et al. 2005, Royle et al. 2005). The fundamental concept of occupancy is enforced through a series of assumptions that can constrain study design and reduce feasibility of occupancy analysis with certain species. Bats, as a group, are difficult to study in an occupancy framework for a variety of

reasons. Most bat research occurs during summer months when bats are actively foraging and caring for flightless pups. These pups grow throughout the summer until late July or early August when they emerge as volant, echolocating individuals. This violates a critical assumption of occupancy that the system population remains closed as the population increases substantially midway through the sampling season. Violation of the closure assumption causes an overestimation of population occupancy. Similarly, northeastern bat species have large home ranges often averaging over 100 hectares (Owen et al. 2003, Menzel et al. 2005). Flight allows these species to travel long distances to optimal foraging sites and back to roosts in one night. Land managers are interested in bat use at a spatial scale that is useful for management, but sampling at these small scales often violates the assumption of independent detections from bats travelling across the landscape. These critical assumption violations make it difficult to assess true occupancy of northeastern bat species. Where these assumptions may be violated, sampling to draw inferences about occupancy may not be feasible or even desirable. However, an occupancy framework can be used to investigate habitat use or site use without the constraint of these assumptions. A habitat use model can account for imperfect detection and provide similar insights into species distribution without the constraint of the closure and independent detection assumptions (MacKenzie 2005, MacKenzie and Bailey 2004).

The goal of this study was to investigate local and landscape conditions associated with habitat use for all bat species present within Cape Cod National Seashore. Based on constraints imposed by acoustic identification to species, habitat use for bat species is defined by both foraging and travelling through the sample space either to foraging or roosting areas. Nine species of insectivorous bat species are known to occur in Massachusetts: northern long-eared bats (*Myotis septentrionalis*), little brown bats (*Myotis lucifugus*), eastern small-footed bats

(*Myotis leibii*), Indiana bats (*Myotis sodalis*), tricolored bats (*Perimyotis subflavus*), big brown bats (*Eptesicus fuscus*), Eastern red bats (*Lasiurus borealis*), hoary bats (*Lasiurus cinereus*), and silver-haired bats (*Lasionycteris noctivagans*). Overall, I predicted that resources and conditions important to species-specific habitat use would be similar to those selected by bat populations in interior areas of the Northeast. I predicted that species with high wing loading and low maneuverability (*E. fuscus, L. borealis, L. cinereus, and L. noctivagans*) would use more open areas and be more likely to use sites proximate to major roadways for ease of travel. Conversely, I predicted that species with low wing loading and high maneuverability (*M. septentrionalis*, *M. lucifugus*, *M. sodalis*, and *M. leibii*) would use more cluttered vegetation types, such as interior forest. Bat species known to use dead trees for diurnal roosting in summer (*Myotis septentrionalis* and *Myotis sodalis*) should chose sites with a higher density of snags that could serve as potential roosts. Furthermore, I predicted that all bat species would increase site use based on proximity to freshwater sources. I expected that average daily wind speed and average daily precipitation would be inversely related to probability of detection for all bat species due to associated difficulties with flying in adverse weather and potential issues with microphone sensitivity.

METHODS

Study Area

Cape Cod National Seashore is located in Barnstable County, Massachusetts (41˚57'N, 70˚ W). Colloquially known as the Outer Cape, my study area included the entirety of the 176.5 km^2 national seashore and some adjacent residential or commercial areas. The Outer Cape of Cape Cod is the northernmost part of the Atlantic coastal plain, a physiographic region of low

relief. This peninsula experiences summer temperatures between 16.05º C to 24.5º C with an average of 27 cm of precipitation. Winters are generally milder than in other areas of the Northeastern US with temperatures ranging from -3.22º C to 4.94º C and an average snowfall of 30.66 cm. Cape Cod National Seashore is characterized by a variety of terrestrial and marine ecosystems including beaches, salt marshes, kettle ponds, and vernal pools. The forested landscape is dominated by pitch pine (*Pinus rigida*), white oak (*Quercus alba*), black oak (*Quercus velutina*), and scrub oak forests (*Quercus ilicifolia*), but also includes heathlands, dunes, and sandplain grasslands. In 2016, Cape Cod National Seashore recorded 4,692,796 recreational visitors and has an annual average of 4,498,222 visitors. This area is home to some of the most popular beaches in the United States including Nauset Lighthouse Beach and Coast Guard Beach (National Oceanic and Atmospheric Administration, National Climatic Data Center, http://www.ncdc.noaa.gov).

Site Selection

Due to the exploratory nature of this study, I chose to randomly acoustically sample throughout Cape Cod National Seashore to collect a representative and unbiased sample of bat species assemblage and habitat use. To select sample sites, I used a map of vegetation types classified by the National Park Service. I excluded marine shore areas and areas identified as highly developed where acoustic detectors could not be safely deployed. From this remaining area, I compiled land cover types into eight vegetation types thought to be meaningful to bat habitat use: coastal plain pondshore/swamp, pitch pine forest, scrub oak forest, dune shrub/heathland/grassland, oak/beech forest, black locust/cedar swamp, developed/disturbed, and red cedar/salt shrub forest.

To capture the range of landscape conditions across the peninsula and ensure adequate spatial interspersion of sites across a north-south gradient, I broke my study area into 3 equal bands: north, middle, and south portions. Then, I placed a 300-m by 300-m grid across the sampling frame to identify a set of prospective sample sites in each band. I set the minimum distance between sample sites at 300 m based on the cone of detection for acoustic detectors (30 m to 100 m depending on vegetation cover conditions). This distance was sufficient to ensure detections at one sample site were not simultaneously recorded as detections at an adjacent site. I numbered all potential sample sites sequentially and determined sampling order using a random number generator. To begin, I randomly selected 2 sample sites in each vegetation category and each directional band to ensure coverage across the north-south gradient. I used stratified random sampling by vegetation type to select remaining sample sites across the entire study area.

Acoustic Sampling

Acoustic detectors have become a mainstay for studying bat ecology due to their ability to detect species that often avoid mist nets (Murray et al. 1999, Coleman et al. 2014). I conducted acoustic sampling at 147 sample sites on Cape Cod National Seashore from 3 June to 20 July 2015 and from 1 June to 24 July 2016 (Figure 2.1). These dates were consistent with requirements for presence/absence surveys defined by the U.S. Fish and Wildlife Service (2015). After performing power analyses using estimates of occupancy and probability of detection from Coleman et al. (2014), I determined that I could model an unbiased and precise estimate of use for most species, including *M. septentrionalis,* by sampling 147 sites 4 times each. I chose to sample at a high number of sites with the understanding that probability of habitat use may be low for bat species in this area and that numerous sample sites would provide a better estimate of

use than numerous replicates (MacKenzie and Royle 2005). I deployed Wildlife Acoustics Song Meter SM3BAT acoustic detectors for 2 consecutive nights at each sample site, on 2 sampling occasions separated by at least 3 weeks (Britzke et al. 2013, Kaiser and O'Keefe 2015). Directional microphones were weatherproofed using the provided foam shield and mounted on 3-m metal poles (Weller and Zaber 2002). SM3BAT detectors are designed to be triggered by audio within a certain frequency range and begin recording. These devices record full-spectrum echolocation recordings that include frequency, length of time, and amplitude information. I programmed each acoustic detector to automatically record echolocations within a 16 kHz to 192 kHz frequency range designed to capture calls of northeastern bat species. Each acoustic detector ran nightly for approximately 12 hours from sunset (1900 EST) to sunrise (700 EST) to capture the full extent of the foraging and travelling period.

Landscape, Local, and Event Metrics

At each sample site, I quantified local vegetation characteristics by sampling vegetation along four 25-m transects arrayed in an "+" pattern around the detector with each transect 90º from adjacent transects. I determined the first transect placement by choosing a random direction using a compass. At 5-m intervals along each transect, I measured understory structure by counting number of woody stems in four 0.5-m height increments using a Robel pole. I measured canopy cover using a spherical densiometer and calculated percent open canopy at each sample site by averaging all 20 readings taken. I also recorded number of living trees and number of dead trees (snags) within a 2-m radius of the current sampling point along the transect. Finally, I noted tree species, diameter at breast height (DBH), total height, and status (alive or dead) of the nearest tree. I used ArcGIS to measure distance to water, distance to human structures such as

houses and buildings, and distance to main roads and dirt roads from each sample point. Following each deployment event, I recorded Julian date (1 through 53 beginning on June 1st), maximum daily temperature (ºC), minimum daily temperature (ºC), average daily humidity (%), average daily wind speed (MPH), and average daily precipitation (cm).

Acoustic Data Processing

I used SonoBat version 3.2.2 software (Arcata, CA, USA) to automatically classify collected echolocation call files to species. I manually vetted all echolocation passes to confirm species identification, reduce bias, and increase identification precision (O'Farrell et al 1999). Although acoustic detectors are an important method, they are not without limitations. Misidentification rates can range from 5% to 30% and may never be eliminated among species with similar call structure, such as myotis species (Armitage and Ober 2010, Barclay 1999, Britzke et al. 2011). By manually vetting all automatic call classification, I mitigated the requirement of an estimate of software uncertainty within my analysis. I categorized bat species detection at a sample site as probable detection or no detection based on quantity and quality of call files. I classified a clear, loud echolocation call with at least 3 passes as good echolocation calls (O'Farrell et al. 1999). Acceptable echolocation calls were lower volume and quality calls that included at least 3 passes and were identified to species by SonoBat. I categorized echolocation calls that were identified to species using SonoBat but had less than 3 passes as poor calls. At least one good or acceptable quality call for a species within a single sampling occasion indicated a probable detection (1) in that location. I classified poor quality calls or an absence of calls as not detected (0) in that site during that sampling occasion.

Model Selection

I selected covariates based on published literature describing bat habitat use in inland regions of the Northeast. These covariates were divided into four categories: landscape-level conditions, local-level conditions, temporal variables, and vegetation type (Table 2.1). For all covariates, I tested for multicollinearity using Pearson's correlation coefficient and did not allow any two highly correlated variables to be used in the same model ($r > 0.60$) to avoid multicollinearity. All covariates were standardized for model selection. For all species which had sufficient detections, I tested 12 *a priori* probability of detection (p) models and determined the most parsimonious model for each species prior to evaluating probability of use models (Table 2.2). I also modeled vegetation type as a function of habitat use (ψ) and determined the most parsimonious model prior to evaluating the landscape- and local-level conditions associated with use to reduce overall number of variables being modeled. I created 17 *a priori* hypotheses to address which local and landscape conditions would be associated with use (Table 2.3). I performed single-species, single-season occupancy models using package unmarked in R (Fiske and Chandler 2011). Each night sampled constituted 1 sampling event with a total of 4 per site. For each species individually, I ranked candidate models using Akaike's Information Criterion (AIC) and assessed goodness-of-fit for the best model. For some species, goodness-of-fit tests indicated poor model fitting and I compared subsequent models using quasi-Akaike's Information Criterion (QAIC) that adjusts for overdispersion. I compared all other models using Akaike's Information Criterion adjusted for small sample sizes (AICc). For species with more than one model with $\Delta AIC < 2$, I model averaged beta coefficients for the top models.

RESULTS

I classified 4,695 call files to species (22.7%) in 2015-2016 from 147 sample sites (Figure 2.2). From these, I identified eight of nine species previously detected in Massachusetts: eastern small-footed bats (*Myotis leibii*), northern long-eared bats (*Myotis septentrionalis*), little brown bats (*Myotis lucifugus*), tricolored bats (*Perimyotis subflavus*), red bats (*Lasiurus borealis*), big brown bats (*Eptesicus fuscus*), silver-haired bats (*Lasionycteris noctivagans*), and hoary bats (*Lasiurus cinereus*).

Myotis septentrionalis Habitat Use

Naïve occupancy for *M. septentrionalis* was 0.109. I detected *M. septentrionalis* most often in the mixed forest vegetation type (n = 6). I detected *M. septentrionalis* during at least one night within at least one sample site from each vegetation category. Increasing average daily precipitation ($\hat{\beta} = 0.816 \pm 0.334, P = 0.015$) and increasing Julian date ($\hat{\beta} = 0.279 \pm 0.445, P = 0.531$) were positively correlated with probability of detection. Increasing average daily wind speed ($\hat{\beta} = -0.289 \pm 0.448, P = 0.519$), minimum daily temperature ($\hat{\beta} = -0.897 \pm 0.533, P = 0.093$), and daily humidity ($\hat{\beta} = -0.642 \pm 0.296, P = 0.030$) were negatively correlated with probability of detection for *M. septentrionalis.* The best model for vegetation type showed a negative association between *M. septentrionalis* and pitch pine forests ($\hat{\beta} = -3.13 \pm 21.74, P = 0.886$). The two most parsimonious single-species, single-season habitat use models (ΔAICc < 2) indicated that distance to freshwater and survey year were associated with habitat use for *M. septentrionalis* (Table 2.4). As distance to water increased, probability of habitat use decreased ($\hat{\beta} = -0.933 \pm 0.516, P = 0.070$). Probability of habitat use for *M. septentrionalis* was lower in 2016 than in 2015 (Figure 2.3). Chi^2 goodness-of-fit test

and $\hat{c}$ value indicated moderate model fit with some underdispersion ($\chi^2 = 16.512, P = 0.500, \hat{c} = 0.9$).

Eptesicus fuscus Habitat Use

Naïve occupancy for *E. fuscus* was 0.741. I detected *E. fuscus* in all eight vegetation categories, but most often in coastal plain pondshore/swamp (n = 16), heathland/grassland (n = 17), and developed/disturbed sites (n = 17). Increasing average daily precipitation ($\hat{\beta} = 0.229 \pm 0.131, P = 0.079$), minimum temperature ($\hat{\beta} = 0.448 \pm 0.175, P = 0.001$), and increasing Julian date ($\hat{\beta} = 0.398 \pm 0.146, P = 0.006$) were positively correlated with probability of detection. Increasing average daily wind speed ($\hat{\beta} = -0.424 \pm 0.130, P = 0.773$) and average daily humidity ($\hat{\beta} = -0.131 \pm 0.121, P = 0.277$) were negatively associated with probability of detection for *E. fuscus.* The null model for vegetation type was the most parsimonious. The two most parsimonious single-species, single-season habitat use models (ΔQAICc < 2) indicated that percent canopy openness and the interaction between canopy openness and vertical structure at a sample location were associated with habitat use for *E. fuscus* (Table 2.5). As percent canopy openness increased, probability of habitat use increased ($\hat{\beta} = 2.45 \pm 1.06, 95\% CI = 0.37, 4.53$). Probability of site use for *E. fuscus* decreased with increasing vertical structure ($\hat{\beta} = -0.508 \pm 0.327, P = 0.120$). Chi^2 goodness-of-fit test and $\hat{c}$ value indicated overdispersion ($\chi^2 = 39.188, P < 0.001, \hat{c} = 2.18$).

Lasiurus borealis Habitat Use

Naïve occupancy for *L. borealis* was 0.381. I positively identified and detected *L. borealis* in all eight vegetation categories, but most often in sites classified as coastal plain

pondshore/swamp (n = 10), scrub oak forest (n = 9), and heathland/grassland (n = 9). The best model for vegetation type showed a negative association between *L. borealis* and oak and beech (mixed) forests ($\hat{\beta} = -0.619 \pm 0.271, P = 0.023$). The most parsimonious model indicated that daily minimum temperature was associated with probability of detection for *L. borealis* (Table 2.4). As minimum temperature increased, probability of detection increased ($\hat{\beta} = 0.490 \pm 0.168, P = 0.035$). There was no single parsimonious model for habitat use of *L. borealis.* The model averaged estimates for the top seven single-species, single-season models indicated *L. borealis* use was positively associated with number of trees ($\hat{\beta} = 0.12 \pm 0.25, 95\% CI = -0.38, 0.61$), number of snags ($\hat{\beta} = 0.61 \pm 0.36, 95\% CI = -0.09, 1.32$), canopy openness ($\hat{\beta} = 0.52 \pm 0.34, 95\% CI = -0.14, 1.19$), and number of stems ($\hat{\beta} = 0.52 \pm 0.25, 95\% CI = 0.03, 1.02$). *L. borealis* use was negatively associated with diameter at breast height of trees and snags in the area ($\hat{\beta} = -0.71 \pm 0.29, 95\% CI = -1.27, -0.15$) and distance to freshwater sources ($\hat{\beta} = -0.43 \pm 0.23, 95\% CI = -0.88, 0.02$). Chi^2 goodness-of-fit test and $\hat{c}$ value indicated overdispersion ($\chi^2 = 27.400, P = 0.12, \hat{c} = 1.46$).

***Lasiurus cinereus* Habitat Use**

Naïve occupancy for *L. cinereus* was 0.558. I detected *L. cinereus* in each vegetation category, but most often in sites classified as coastal plain pondshore/swamp (n = 16). Increasing daily maximum temperature was positively associated with probability of detection for *L. cinereus* ($\hat{\beta} = 0.928 \pm 0.172, P < 0.001$). The best model for vegetation type showed a negative association between *L. cinereus* and oak and beech (mixed) forests ($\hat{\beta} = -0.731 \pm 0.247, P = 0.003$). There was no single parsimonious model for habitat use of *L. cinereus.* The model averaged estimates for the top ten single-species, single-season models (ΔQAICc < 2) indicated

that *L. cinereus* use was positively associated with number of snags ($\hat{\beta} = 0.43 \pm 0.34, 95\%\ CI = -0.24, 1.10$), canopy openness ($\hat{\beta} = 0.46 \pm 0.40, 95\%\ CI = -0.32, 1.24$), proximity to manmade structures ($\hat{\beta} = 0.20 \pm 0.30, 95\%\ CI = -0.39, 0.78$), and sample year ($\hat{\beta} = 0.26 \pm 0.25, 95\%\ CI = -0.24, 0.76$). *L. cinereus* use was negatively associated with number of trees ($\hat{\beta} = -0.06 \pm 0.28, 95\%\ CI = -0.61, 0.49$), number of stems ($\hat{\beta} = -0.41 \pm 0.27, 95\%\ CI = -0.94, 0.12$), diameter at breast height of trees and snags ($\hat{\beta} = -0.52 \pm 0.34, 95\%\ CI = -1.18, 0.14$), distance to dirt roads or paths ($\hat{\beta} = -0.23 \pm 0.24, 95\%\ CI = -0.69, 0.23$), and proximity to freshwater ($\hat{\beta} = -0.18 \pm 0.23, 95\%\ CI = -0.63, 0.27$). Chi^2 goodness-of-fit test and $\hat{c}$ value indicated moderate model fit with some underdispersion ($\chi^2 = 15.639, P = 0.6, \hat{c} = 0.86$).

***Lasionycteris noctivagans* Habitat Use**

Naïve occupancy for *L. noctivagans* was 0.741. I detected *L. noctivagans* at least once in each vegetation type, but most often in coastal plain pondshore/swamp (n = 18) and developed/disturbed (n = 17) sites. The most parsimonious model indicated that maximum daily temperature was associated with probability of detection for *L. noctivagans* (Table 2.6). As maximum daily temperature increased, probability of detection increased ($\hat{\beta} = 0.724 \pm 0.124, P < 0.001$). The best (ΔQAIC < 2) two models for vegetation type showed a negative association between *L. noctivagans* and oak and beech (mixed) forests ($\hat{\beta} = -0.462 \pm 0.190, P = 0.002$) and a positive association with coastal plain pondshore/swamp vegetation categories ($\hat{\beta} = 0.982 \pm 1.243, P = 0.429$). No local or landscape covariates were associated with habitat use. Chi^2 goodness-of-fit test and $\hat{c}$ value indicated overdispersion ($\chi^2 = 36.908, P < 0.001, \hat{c} = 2.06$).

Myotis leibii, Myotis lucifugus, and *Perimyotis subflavus* Habitat Use

Due to the low number of detections of *M. leibii*, *M. lucifugus*, and *P. subflavus*, I was unable to assess habitat use using occupancy models. Out of 147 sample sites, I detected *M. leibii* on 2 sampling occasion at 2 sample sites classified as red cedar/salt shrub forest and black locust vegetation types. I positively identified *M. lucifugus* at 1 sample site categorized as scrub oak forest over 3 sampling events. *M. lucifugus* was only detected in 2015. I detected *P. subflavus* at 6 sample sites during 9 sampling events. Events with *P. subflavus* detections were at sites within coastal plain pondshore/swamp (n = 3), and developed/disturbed (n = 3) vegetation types.

DISCUSSION

This research is the first to document populations of *M. septentrionalis, L. borealis, E. fuscus, L. noctivagans,* and *L. cinereus* within Cape Cod National Seashore in over 100 years. Populations of *M. leibii*, *M. lucifugus*, and *P. subflavus* may also be present, but more intensive research focused on these three species should be conducted. These habitat use models indicate that bats in the northeastern Atlantic coastal plain make decisions to use resources and conditions at the landscape- and local-level similar to populations in the interior Northeast. *E. fuscus* used sites associated with characteristics we would expect based on their large body size and high wing loading, such as sites with more canopy openness and low vertical structure. *E. fuscus* also showed their adaptability to this highly developed, coastal landscape through a high number of detections in sites categorized as coastal plain pondshore and developed or disturbed vegetation. The three migratory tree bats (*L. borealis, L. cinereus*, and *L. noctivagans*) all showed an

increased probability of detection with increasing temperature, possibly due to an influx of migrants moving northward as summer progressed. The significant negative association between site use and oak and beech forest for *L. borealis, L. cinereus,* and *L. noctivagans* corresponds to what we would expect from large-bodied bats with high wing loading and low maneuverability that tend to use more open environments. These three species were most often detected in coastal plain pondshore and swamp vegetation highlighting their adaptability to this coastal ecosystem.

Large population declines across most of its range and recent federal and state-level listing make *M. septentrionalis* a species of special concern and interest. I hypothesized that probability of habitat use by *M. septentrionalis,* as a forest interior specialist, would be associated with low canopy openness, high number of snags, and large distance from main roads. I hypothesized that *M. septentrionalis,* like all bat species, would use sites close to fresh water sources. By using mostly mixed forest sites and avoiding flight during periods of high wind or high humidity, *M. septentrionalis* used sites with conditions we would expect based on prior studies from inland areas of the Northeast. However, instead of these mixed forest areas showing a significant positive association with *M. septentrionalis* use, the negative association with pitch pine forests showed the strongest support. This vegetation category is dominant within Cape Cod National Seashore, but infrequent in interior areas of the Northeast. Future work should investigate insect abundance and in-depth diurnal roost assessment to determine the cause of this negative association.

Large differences in probability of habitat use between sample years for *M. septentrionalis* imply potentially concerning population declines. During summer of 2016, the Northeast experienced the largest outbreak of gypsy moths (*Lymantria dispar dispar*) on the Cape since 1981. Forests were defoliated during peak summer months, potentially reducing

insect populations in the area. The negative correlation between survey year and probability of habitat use by *M. septentrionalis* may have been due to negative effects caused by gypsy moths on insect abundance or forest cover. Although this gypsy moth infestation may have affected habitat use during summer of 2016, this study was unable to determine what factors played a role in the interannual difference. White-nose syndrome, for example, may have reduced the population in this area, with effects becoming recognizable in 2016. Continued annual investigation into habitat use and abundance of *M. septentrionalis* on Cape Cod National Seashore is needed to determine the effect of gypsy moth defoliations on populations of this and other species of interest. Compared to all other *Myotis* species, *M. septentrionalis* were detected at a relatively high number of sample sites during this sampling period. This contrasts from research conducted in the southeastern Atlantic coastal plain that found that *Myotis* bats were unlikely to use coastal plain ecosystems for foraging and roosting (Furlonger et al. 1987, Menzel et al. 2000). To date, the effects of white-nose syndrome are more prominent in populations of northeastern bat species than southeastern species. Consequently, southeastern *Myotis* species or populations that have yet to be affected by white-nose syndrome may behave differently in the Atlantic coastal plain ecosystem than northeastern *Myotis* species.

Habitat use models provide concrete species distribution information and insight into vegetation types and important local, landscape, and temporal attributes that affect probability of site use and detection by bats in the northeastern Atlantic coastal plain. With confirmed presence of these eight species, park managers can focus management efforts on species of interest. The three migratory tree bat species (*L. borealis*, *L. cinereus*, and *L. noctivagans*) were positively detected throughout the entire sampling period (1 June – 24 July) indicating potential resident populations. Park managers interested in promoting *M. septentrionalis* use within Cape Cod

National Seashore should focus on maintaining the wide variety of kettle ponds, particularly within oak and beech mixed forest vegetation, to provide access to freshwater sources. Management actions within pitch pine forests should not be designed with *M. septentrionalis* in mind. However, avoidance of pitch pine vegetation may assist with manager's compliance of the Endangered Species Act as prescribed burning actions to maintain heathlands or pitch pine dominant areas are undertaken. Although *M. septentrionalis* are not likely to be roosting in the area, to fully satisfy the Endangered Species Act requirement, short-term monitoring prior to any prescribed burns should be conducted to avoid mortality (Dickinson et al. 2010). With updated information about bats within Cape Cod National Seashore, the National Park Service can begin to focus efforts on vegetation types and specific sample sites that are important to declining bat species and continue to monitor species assemblage and distribution.

Although an important tool for biologists studying bat assemblage and habitat use, acoustic detectors have limitations including inability to accurately sex or count individuals, issues with acoustically cryptic species (Britzke et al. 2013), and plasticity of echolocation call structure (Barclay 1999). Model fitting with both underdispersion and overdispersion for all species identified indicate heterogeneity that has been unaccounted for or a lack of power to draw inferences. Future research within Cape Cod National Seashore should use a combination of acoustic identification methods and capture methods and ensure power to draw inferences to build off of this important baseline information.

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Table 2.1 All possible sample site-specific landscape, local, and temporal variables considered in single-species, single-season occupancy models for all bat species acoustically detected within Cape Cod National Seashore, 2015-2016.

Covariates Associated with Probability of Habitat Use (ψ)	
Landscape Covariates	Description
Water	Euclidean distance to the nearest source of fresh water (m)
Road	Euclidean distance to the nearest major roadway (m)
Structure	Euclidean distance to the nearest manmade structure (m)
Dirt	Euclidean distance to the nearest dirt road or path (m)
Local Variable	**Description**
Canopy	Average overstory canopy openness (%)
Snags	Average number of dead trees (snags) per m^2
Trees	Average number of living trees per m^2
DBH	Average diameter at breast height of trees within sample site (cm)
Vegetation Type Variable	**Description**
VegA	Vegetation type A: coastal plain pondshore and swamp
VegB	Vegetation type B: pitch pine forest
VegC	Vegetation type C: scrub oak forest
VegD	Vegetation type D: dune shrub, heathland, and grassland
VegE	Vegetation type E: oak and beech (mixed) forest
VegF	Vegetation type F: black locust and cedar swamp
VegG	Vegetation type G: developed and disturbed
VegH	Vegetation type H: red cedar and salt shrub forest
Temporal Variable	**Description**
Year	Sample year
Covariates Associated with Probability of Detection (p)	
Detection Variable	Description
Precip	Average daily precipitation (cm)
Wind	Average daily wind speed (mph)
MaxT	Maximum daily temperature (ºC)
MinT	Minimum daily temperature (ºC)
Humid	Average daily humidity (%)
Date	Julian date: continuous count of days beginning at June 1st (1) and ending on July 24th (53)

Table 2.2 List of 12 probability of detection (*p*) models, the covariates included with descriptions, and the hypothesized effect for the single-species, single-season occupancy models for all bat species acoustically detected within Cape Cod National Seashore, 2015-2016.

Model	Covariate names and descriptions	Hypothesized Effect
Wind	Wind: Average daily wind speed	– Reduces foraging ability
Maximum temperature	MaxT: Maximum daily temperature	+ Increases foraging ability
Minimum temperature	MinT: Minimum daily temperature	– Reduces foraging ability
Precipitation	Precip: Average daily precipitation	– Reduces foraging ability
Humidity	Humid: Average daily humidity	– Reduces foraging ability
Julian date	Date: Julian date	+ Increases foraging ability
Vertical structure	Stems: estimate of vertical structure at the sample site	+ or – depending on morphology
Canopy openness	Canopy: estimate of canopy openness at the sample site	+ or – depending on morphology
Succession stage	Stems+Canopy: vertical structure and canopy openness interaction	+ or – depending on morphology
Global without maximum temperature	Wind+MinT+Precip+Humid+Date+Stems+Canopy: maximum temperature removed due to high correlation with minimum temperature	No hypothesis
Global without minimum temperature	Wind+MaxT+Precip+Humid+Date+Stems+Canopy: minimum temperature removed due to high correlation with maximum temperature	No hypothesis
Null	No covariates included	No hypothesis

Table 2.3 List of 17 probability of habitat use (ψ) models, the covariates included with descriptions, and the hypothesized effect for the single-species, single-season occupancy models for all bat species acoustically detected within Cape Cod National Seashore, 2015-2016.

Model	Landscape covariate names and descriptions	Hypothesized effect
Freshwater sources	Water: distance to nearest freshwater source	+ Increases foraging ability and hydration
Manmade structure	Structure: distance to nearest manmade structure	+ or – depending on roosting needs
Dirt roads	Dirt: distance to nearest dirt road or path	+ Increases efficient travel for foraging
Main roads	Road: distance to nearest major roadway	+ or – depending on roosting needs and sensitivity to disturbance
Sample year	Year: sample year	No change
Human Disturbance	Structure+Road: distance to nearest major roadway and manmade structure interaction	– Sensitivity to disturbance
Global landscape variables	Water+Structure+Dirt+Road+Year	No hypothesis
Model	**Local covariate names and descriptions**	**Hypothesized**
Live tree roost availability	Trees: average number of living trees at sample site	+ or – depending on roost needs
Dead tree roost availability	Snags: average number of dead trees at sample site	+ or – depending on roost needs
Midstory structure and roost availability	DBH: average diameter at breast height for trees at sample site	+ or – depending on morphology and roost needs
Vertical structure	Stems: estimate of vertical structure at the sample site	+ or – depending on morphology
Canopy openness	Canopy: estimate of canopy openness at the sample site	+ or – depending on morphology
Succession stage	Stems+Canopy: vertical structure and canopy openness interaction	+ or – depending on morphology
Global local variables without canopy	Trees+Snags+DBH: canopy removed due to high correlation with DBH	No hypothesis
Global local variables without DBH	Trees+Snags+Canopy: DBH removed due to high correlation with canopy	No hypothesis
Probability of detection	Includes best covariate from probability of detection models	See hypotheses in Table 2.2
Null	No covariates included	No hypothesis

Table 2.4 List of single-species, single-season habitat use models for *M. septentrionalis*. Includes model name, number of parameters (k), Akaike's Information Criterion adjusted for small sample sizes (AICc), change in AICc relative to the best model, and cumulative weight of each subsequent model.

Model Name	K	AICc	ΔAICc	cum. ω_i
ψ(Water+VegB), *p*(Wind+MinT+Precip+Humid+Date)	9	149.58	0.00	0.32
ψ(Year+VegB), *p*(Wind+MinT+Precip+Humid+Date)	9	149.92	0.34	0.59
ψ(Water+Structure+Dirt+Road+Year+VegB), *p*(Wind+MinT+Precip+Humid+Date)	13	152.52	2.94	0.66
ψ(Dirt+VegB), *p*(Wind+MinT+Precip+Humid+Date)	9	153.13	3.55	0.72
ψ(.), *p*(Wind+MinT+Precip+Humid+Date)	7	153.70	4.12	0.76
ψ(Stems+VegB), *p*(Wind+MinT+Precip+Humid+Date)	9	153.96	4.38	0.79
ψ(Canopy+VegB), *p*(Wind+MinT+Precip+Humid+Date)	9	154.22	4.64	0.82
ψ(DBH+VegB), *p*(Wind+MinT+Precip+Humid+Date)	9	154.32	4.73	0.85
ψ(Structure+VegB), *p*(Wind+MinT+Precip+Humid+Date)	9	154.34	4.75	0.88
ψ(Trees+VegB), *p*(Wind+MinT+Precip+Humid+Date)	9	154.36	4.78	0.91
ψ(Road+VegB), *p*(Wind+MinT+Precip+Humid+Date)	9	154.42	4.84	0.94
ψ(Snags+VegB), *p*(Wind+MinT+Precip+Humid+Date)	9	154.44	4.86	0.97

Table 2.5 List of single-species, single-season habitat use models for *E. fuscus*. Includes model name, number of parameters (k), Akaike's Information Criterion adjusted for overdispersion (QAIC), change in QAIC relative to the best model, and cumulative weight of each subsequent model.

Model Name	K	QAICc	ΔQAICc	cum. ω_i
ψ(Canopy), *p*(Wind+MinT+Precip+Humid+Date)	9	316.40	0.00	0.42
ψ(Stems+Canopy), *p*(Wind+MinT+Precip+Humid+Date)	10	317.72	1.32	0.64
ψ(DBH), *p*(Wind+MinT+Precip+Humid+Date)	9	319.21	2.81	0.74
ψ(Canopy+Trees+Snags), *p*(Wind+MinT+Precip+Humid+Date)	11	320.21	3.82	0.81
ψ(.), *p*(Wind+MinT+Precip+Humid+Date)	8	321.00	4.60	0.85
ψ(Stems), *p*(Wind+MinT+Precip+Humid+Date)	9	322.17	5.77	0.87
ψ(Dirt), *p*(Wind+MinT+Precip+Humid+Date)	9	322.45	6.05	0.89
ψ(Water), *p*(Wind+MinT+Precip+Humid+Date)	9	322.59	6.19	0.91
ψ(Trees+Snags+DBH), *p*(Wind+MinT+Precip+Humid+Date)	11	323.05	6.65	0.93
ψ(Trees), *p*(Wind+MinT+Precip+Humid+Date)	9	323.18	6.78	0.94
ψ(Year), *p*(Wind+MinT+Precip+Humid+Date)	9	323.25	6.85	0.95

Table 2.6 List of single-species, single-season habitat use models for *L. borealis*. Includes model name, number of parameters (k), Akaike's Information Criterion adjusted for overdispersion (QAIC), change in QAIC relative to the best model, and cumulative weight of each subsequent model.

Model Name	K	QAICc	ΔQAICc	cum. ω_i
ψ(Trees+Snags+DBH+VegE), *p*(MinT)	8	294.08	0.00	0.15
ψ(DBH+VegE), *p*(MinT)	6	294.27	0.19	0.29
ψ(Stems+VegE), *p*(MinT)	6	294.43	0.36	0.42
ψ(Water+VegE), *p*(MinT)	6	295.04	0.96	0.52
ψ(Stems+Canopy+VegE), *p*(MinT)	7	295.23	1.15	0.60
ψ(Canopy+Trees+Snags+VegE), *p*(MinT)	8	295.93	1.86	0.66
ψ(Snags+VegE), *p*(MinT)	6	295.99	1.92	0.72
ψ(Dirt+VegE), *p*(MinT)	6	296.09	2.01	0.78
ψ(Structure+VegE), *p*(MinT)	6	296.19	2.11	0.83
ψ(Canopy+VegE), *p*(MinT)	6	296.56	2.49	0.88
ψ(Year+VegE), *p*(MinT)	6	296.99	2.91	0.91
ψ(Trees+VegE), *p*(MinT)	6	297.37	3.29	0.94
ψ(Road+VegE), *p*(MinT)	6	297.79	3.72	0.96

Table 2.7 List of single-species, single-season habitat use models for *L. cinereus*. Includes model name, number of parameters (k), Akaike's Information Criterion adjusted for small sample sizes (AICc), change in AICc relative to the best model, and cumulative weight of each subsequent model.

Model Name	K	AICc	ΔAICc	cum. ω_i
ψ(DBH+VegE), *p*(MaxT)	5	487.10	0.00	0.14
ψ(Stem+VegE), *p*(MaxT)	5	487.34	0.24	0.26
ψ(Stems+Canopy+VegE), *p*(MaxT)	6	487.78	0.68	0.35
ψ(Snags+VegE), *p*(MaxT)	5	488.05	0.96	0.44
ψ(Trees+Snags+DBH+VegE), *p*(MaxT)	7	488.06	0.97	0.52
ψ(Canopy+VegE), *p*(MaxT)	5	488.21	1.12	0.60
ψ(Dirt+VegE), *p*(MaxT)	5	488.48	1.38	0.66
ψ(Year+VegE), *p*(MaxT)	5	488.48	1.38	0.73
ψ(Water+VegE), *p*(MaxT)	5	488.98	1.88	0.79
ψ(Structure+VegE), *p*(MaxT)	5	489.00	1.90	0.84
ψ(Road+VegE), *p*(MaxT)	5	489.41	2.31	0.88
ψ(Trees+VegE), *p*(MaxT)	5	489.55	2.45	0.92
ψ(Canopy+Trees+Snags+VegE), *p*(MaxT)	7	489.63	2.53	0.96

Table 2.8 List of single-species, single-season season habitat use models for *L. noctivagans*. Includes model name, number of parameters (k), Akaike's Information Criterion adjusted for overdispersion (QAIC), change in QAIC relative to the best model, and cumulative weight of each subsequent model.

Model Name	K	QAICc	ΔQAICc	cum. ω_i
ψ(.), *p*(MaxT)	4	328.48	0.00	0.27
ψ(DBH+VegA+VegE+VegF), *p*(MaxT)	8	330.83	2.35	0.35
ψ(Stems+VegA+VegE+VegF), *p*(MaxT)	8	330.84	2.36	0.43
ψ(Water+VegA+VegE+VegF), *p*(MaxT)	8	331.37	2.89	0.49
ψ(Road+VegA+VegE+VegF), *p*(MaxT)	8	331.48	3.00	0.55
ψ(Canopy+VegA+VegE+VegF), *p*(MaxT)	8	331.48	3.00	0.61
ψ(Trees+VegA+VegE+VegF), *p*(MaxT)	8	331.51	3.03	0.67
ψ(Year+VegA+VegE+VegF), *p*(MaxT)	8	331.54	3.06	0.73
ψ(Structure+VegA+VegE+VegF), *p*(MaxT)	8	331.65	3.17	0.78
ψ(Road+Structure+VegA+VegE+VegF), *p*(MaxT)	9	331.66	3.18	0.84
ψ(Dirt+VegA+VegE+VegF), *p*(MaxT)	8	331.94	3.46	0.89
ψ(Stems+Canopy+VegA+VegE+VegF), *p*(MaxT)	9	331.95	3.47	0.93
ψ(Snags+VegA+VegE+VegF), *p*(MaxT)	8	331.96	3.49	0.98

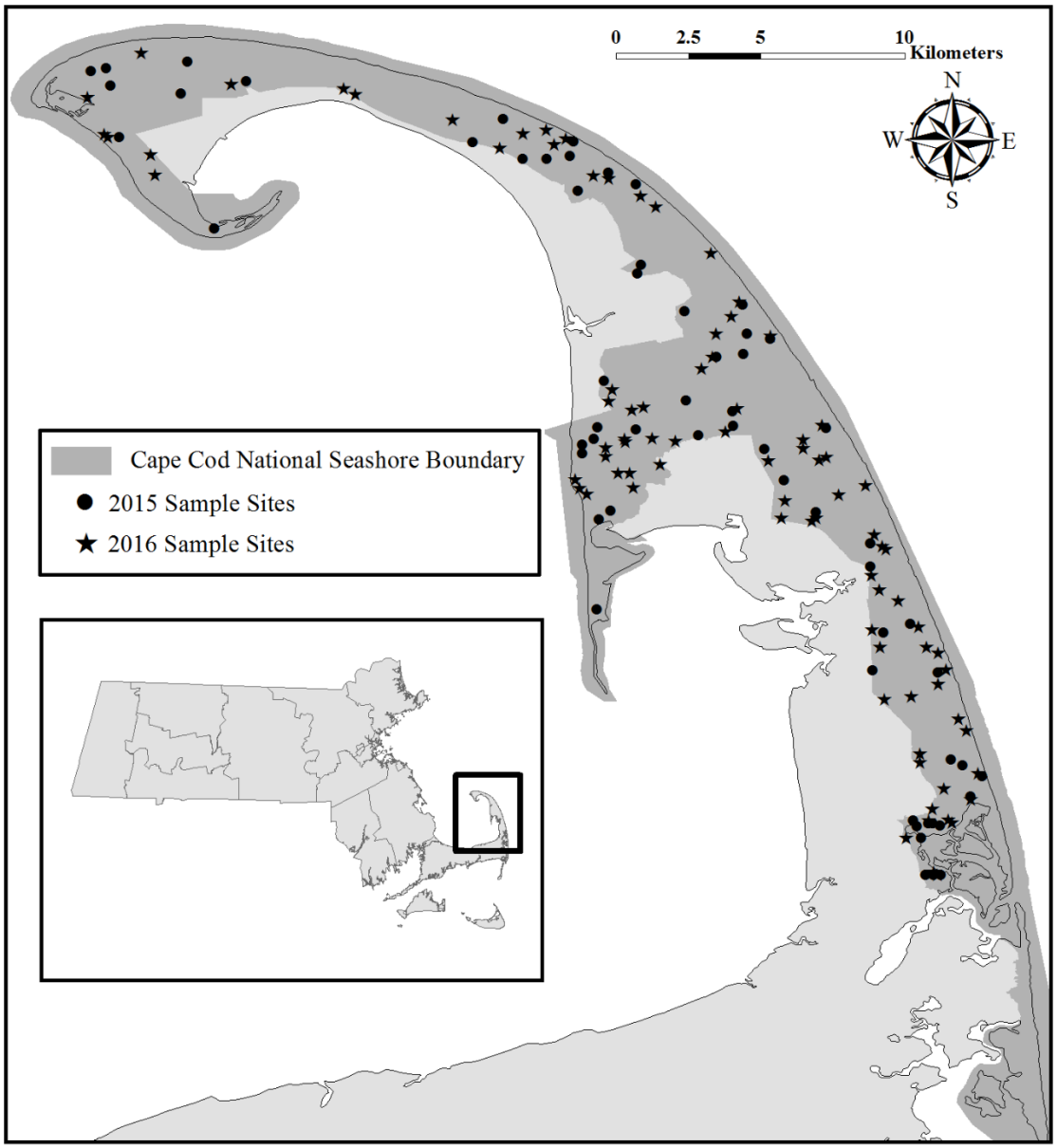

Figure 2.1 Map of study area with location of 147 passive acoustic sampling sites for the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

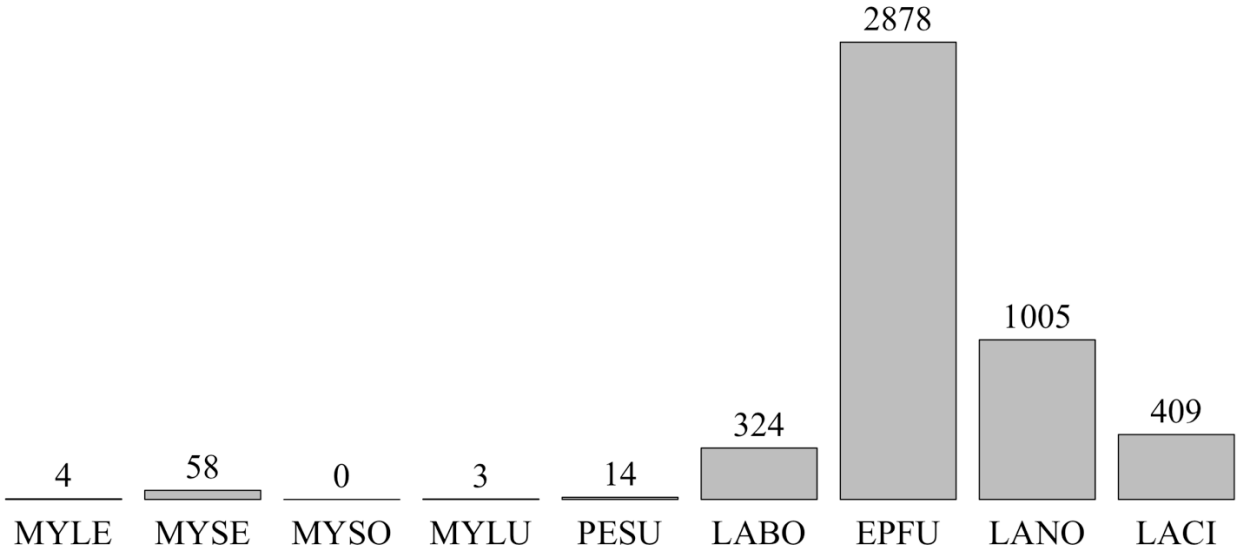

Figure 2.2 Number of full-spectrum acoustic calls classified to species during the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016. MYLE: eastern small-footed bat (*Myotis leibii*), MYSE: northern long-eared bat (*Myotis septentrionalis),* MYSO: Indiana bat (*Myotis sodalis*), MYLU: little brown bat (*Myotis lucifugus*), PESU: tricolored bat (*Perimyotis subflavus*), LABO: eastern red bat (*Lasiurus borealis*), EPFU: big brown bat (*Eptesicus fuscus*), LANO: silver-haired bat (*Lasionycteris noctivagans*), and LACI: hoary bats (*Lasiurus cinereus*).

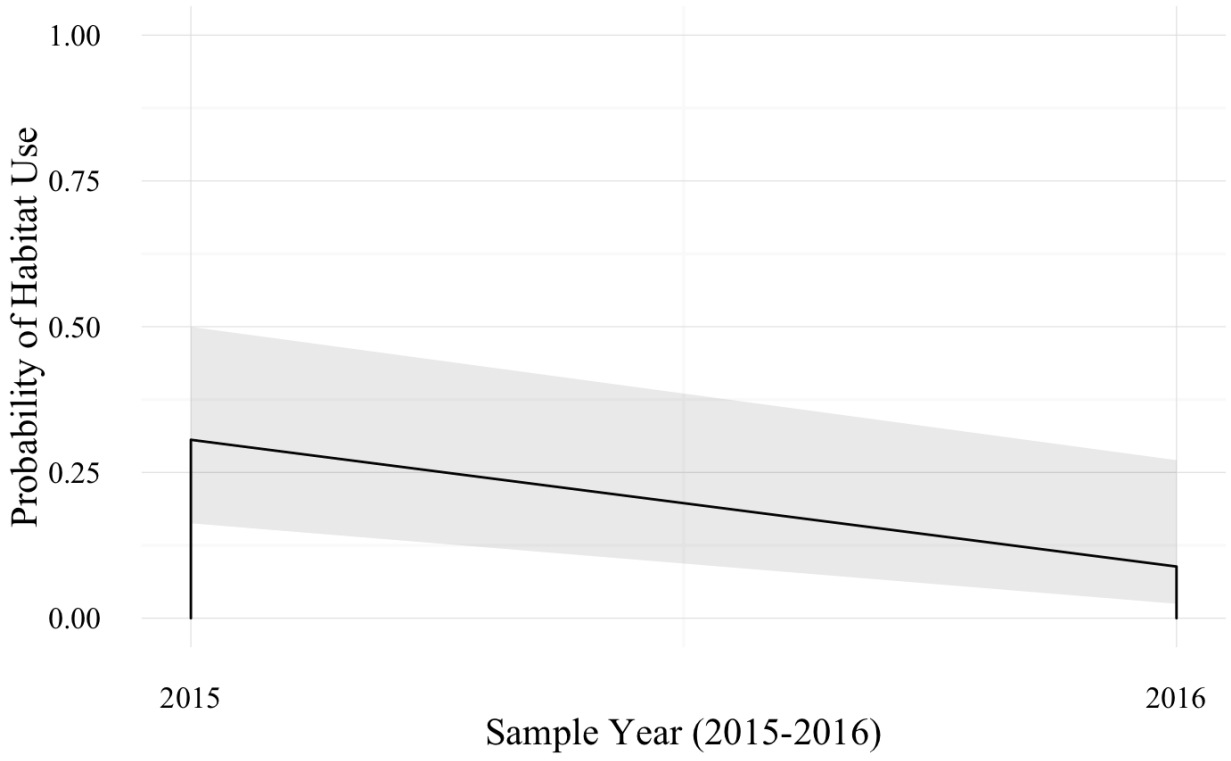

Figure 2.3 Probability of habitat use by northern long-eared bats (*Myotis septentrionalis*) from the best single-species, single-season habitat use model between sample years during the acoustic monitoring conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

CHAPTER 3: ESTIMATION OF CO-OCCURRENCE BETWEEN TWO NORTHEASTERN BAT SPECIES: *EPTESICUS FUSCUS* AND *MYOTIS SEPTENTRIONALIS*

ABSTRACT

The big brown bat (*Eptesicus fuscus*) and the northern long-eared bat (*Myotis septentrionalis*) are two bat species known to co-occur regularly within Cape Cod National Seashore. The effect of co-occurrence is often ignored in studies that focus on factors associated with habitat use for species of interest. To better understand the interspecific conditions associated with habitat use for these two species, I conducted conditional two-species occupancy modelling. My goal was to determine the effect of this co-occurrence on both species, with an emphasis on *M. septentrionalis* as a recently listed species of concern. I included canopy openness and sample year as covariates in my models based on results from single-species, single-season occupancy models for each species. For the conditional two-species occupancy analysis, the two most parsimonious models based on ΔAICc < 2 indicated that *M. septentrionalis* habitat use was higher in sites where *E. fuscus* was present rather than absent ($\psi^{ME} = 0.181$ and $\psi^{Me} = 0.090$). The species interaction factor for the most parsimonious model indicated *M. septentrionalis* use sites independent of *E. fuscus* presence. Managers interested in promoting site use by *M. septentrionalis* should focus efforts on other landscape or local resources and conditions associated with use.

Keywords: Cape Cod National Seashore, conditional two-species occupancy, co-occurrence, *Eptesicus fuscus*, habitat use, *Myotis septentrionalis*

INTRODUCTION

Individuals make habitat selection choices at multiple scales, where landscape-level decisions shape local-level decisions. This makes assessment of habitat use across multiple spatial scales essential for effective management and conservation of target species (Johnson 1980, Orians and Wittenberger 1991). For example, in summer months, bats make landscape- and local-level decisions based on resources and conditions required for successful foraging and adequate diurnal roosting needs. At the landscape scale, most bat species choose landscapes where water sources are available for both hydration and heightened insect abundance for foraging (Barclay 1991). Within a landscape, bats select sites with specific conditions such as suitable roost tree species. Landscape-level selection determines overall distribution of a species, whereas local-level habitat use decisions influence the placement of foraging areas, roosting areas, and home ranges. Beyond landscape and local scales, patterns of habitat use may reflect responses to intraspecific and interspecific interactions in positive, negative, or neutral forms. Co-occurrence interactions can lead to competitive exclusion and niche partitioning in response to competition and competitive displacement (Nicholls and Racey 2006). Conversely, co-occurrence where resources are partitioned sufficiently to prevent competition promotes habitat use independence or species aggregation through intraspecific learning or habitat copying (Valone and Templeton 2002, Matthysen 2005).

For bats, the few studies that examine species interactions and co-occurrence are focused on analyses of diet, echolocation call structure, and morphology as indicators of resource partitioning (Aldridge and Rautenbach 1987, Fenton 1982). Studies investigating bat habitat use primarily focus on local conditions associated with site use, but usually fail to consider potential effects that species interactions have on local habitat use decisions. Investigation of the role that

presence or absence of a heterospecific potential competitor may have on habitat use patterns is underdeveloped for bats as a group (Findley 1993). Work focused solely on local and landscape conditions cannot fully explain factors driving distribution and habitat use. Although it can be difficult to directly observe the influence of interspecific interactions in shaping habitat use, the addition of species interaction as a predictor variable can provide insight into understanding all factors responsible for shaping habitat use.

Larger species competitively displacing smaller species from high quality food-resources or territories is shown in certain groups such as birds and some mustelids, but unproven in others (Martin and Martin 2001, St-Pierre et al. 2006). All insectivorous bat species prefer to forage around areas with standing water that provide habitat for emerging insects (Ford et al. 2002, Kunz and Fenton 2003, Owen et al. 2004). Larger bat species may be able to compete better for high-quality insect resources in these foraging areas by physically displacing smaller competitors, eventually leading to avoidance by smaller species. Many northeastern insectivorous bat species also exploit manmade light sources as foraging opportunities. In these high traffic areas, previous research has shown that larger species competitively displace smaller bat species from foraging opportunities (Furlonger et al. 1987). This competitive aggression toward smaller insectivorous species may drive habitat use and has the potential for reducing fitness of displaced species.

Northern long-eared bats (*Myotis septentrionalis*) are a small, forest interior specialist that have experienced large population declines due to a recently emerging disease called white-nose syndrome. These declines led the U.S. Fish & Wildlife Service to list *M. septentrionalis* as threatened under the Endangered Species Act in 2015. The diet of *M. septentrionalis* relies heavily on Lepidopterans, but also includes Coleopterans, Trichopterans, and Dipterans (Brack

and Whitaker 2001). In contrast, big brown bats (*Eptesicus fuscus*) are a relatively larger, generalist species that have been less affected by white-nose syndrome (Furlonger et al. 1987). They are currently one of the most abundant bat species in the northeastern United States and tend to occupy more open environments due to their large size, high wing loading, and low maneuverability (Kurta and Baker 1990). *E. fuscus* have a diverse and flexible diet throughout their distribution, of which hard-bodied Hemipterans usually make up the majority (Whitaker 1995). Coleoptera and Lepidoptera are also included in the diet of *E. fuscus* in northeastern populations (Brigham and Fenton 1991, Hamilton and Barclay 1998). *E. fuscus* and *M. septentrionalis* overlap in their range and are known to co-occur (Kurta and Baker 1990, Caceres and Barclay 2000). Overlap in distribution and dietary requirements may lead to competition or competitive displacement.

As a federally threatened species, wildlife managers in the northeastern United States are interested in site use by *M. septentrionalis* and potential barriers to recovery. Future management actions must consider effects of potential competitor species and may choose to improve habitat conditions in target areas where competitor species are absent to have the greatest effect on recovery. The goal of this study was to assess how use of an area by *M. septentrionalis* may be positively or negatively associated with use by *E. fuscus*. Based on body size, I expected probability of site use by *M. septentrionalis* to be higher in sites where *E. fuscus* were absent, thereby avoiding the larger competitor. I also predicted that probability of site use of *M. septentrionalis* would be higher in environments with a large amount of vertical structure (i.e. clutter) where *E. fuscus* are less likely to forage due to their large size.

METHODS

Study Area

Cape Cod National Seashore is located in Barnstable County, Massachusetts (41˚57'N, 70˚ W). Colloquially known as the Outer Cape, my study area included the entirety of the 176.5 km^2 national seashore and some adjacent residential or commercial areas. The Outer Cape of Cape Cod is the northernmost part of the Atlantic coastal plain, a physiographic region of low relief. This peninsula experiences summer temperatures between 16.05º C to 24.5º C with an average of 27 cm of precipitation. Winters are generally milder than in other areas of the Northeastern US with temperatures ranging from -3.22º C to 4.94º C and an average snowfall of 30.66 cm. Cape Cod National Seashore is characterized by a variety of terrestrial and marine ecosystems including beaches, salt marshes, kettle ponds, and vernal pools. The forested landscape is dominated by pitch pine (*Pinus rigida*), white oak (*Quercus alba*), black oak (*Quercus velutina*), and scrub oak forests (*Quercus ilicifolia*), but also includes heathlands, dunes, and sandplain grasslands. In 2016, Cape Cod National Seashore recorded 4,692,796 recreational visitors and has an annual average of 4,498,222 visitors. This area is home to some of the most popular beaches in the United States including Nauset Lighthouse Beach and Coast Guard Beach (National Oceanic and Atmospheric Administration, National Climatic Data Center, http://www.ncdc.noaa.gov).

Site Selection

Due to the exploratory nature of this study, I chose to randomly acoustically sample throughout Cape Cod National Seashore to collect a representative and unbiased sample of bat

species assemblage and habitat use. To select sample sites, I used a map of vegetation types classified by the National Park Service. I excluded marine shore areas and areas identified as highly developed where acoustic detectors could not be safely deployed. From this remaining area, I compiled land cover types into eight vegetation types thought to be meaningful to bat habitat use: coastal plain pondshore/swamp, pitch pine forest, scrub oak forest, dune shrub/heathland/grassland, oak/beech forest, black locust/cedar swamp, developed/disturbed, and red cedar/salt shrub forest.

To capture the range of landscape conditions across the peninsula and ensure adequate spatial interspersion of sites across a north-south gradient, I broke my study area into 3 equal bands: north, middle, and south portions. Then, I placed a 300-m by 300-m grid across the sampling frame to identify a set of prospective sample sites in each band. I set the minimum distance between sample sites at 300 m based on the cone of detection for acoustic detectors (30 m to 100 m depending on vegetation cover conditions). This distance was sufficient to ensure detections at one sample site were not simultaneously recorded as detections at an adjacent site. I numbered all potential sample sites sequentially and determined sampling order using a random number generator. To begin, I randomly selected 2 sample sites in each vegetation category and each directional band to ensure coverage across the north-south gradient. I used stratified random sampling by vegetation type to select remaining sample sites across the entire study area.

Acoustic Sampling

Acoustic detectors have become a mainstay for studying bat ecology due to their ability to detect species that often avoid mist nets (Murray et al. 1999, Coleman et al. 2014). I conducted acoustic sampling at 147 sample sites on Cape Cod National Seashore from 3 June to

20 July 2015 and from 1 June to 24 July 2016 (Figure 2.1). These dates were consistent with requirements for presence/absence surveys defined by the U.S. Fish and Wildlife Service (2015). After performing power analyses using estimates of occupancy and probability of detection from Coleman et al. (2014), I determined that I could model an unbiased and precise estimate of use for most species, including *M. septentrionalis,* by sampling 147 sites 4 times each. I chose to sample at a high number of sites with the understanding that probability of habitat use may be low for bat species in this area and that numerous sample sites would provide a better estimate of use than numerous replicates (MacKenzie and Royle 2005). I deployed Wildlife Acoustics Song Meter SM3BAT acoustic detectors for 2 consecutive nights at each sample site, on 2 sampling occasions separated by at least 3 weeks (Britzke et al. 2013, Kaiser and O'Keefe 2015). Directional microphones were weatherproofed using the provided foam shield and mounted on 2-m metal poles (Weller and Zaber 2002). SM3BAT detectors are designed to be triggered by audio within a certain frequency range and begin recording. These devices record full-spectrum echolocation recordings that include frequency, length of time, and amplitude information. I programmed each acoustic detector to automatically record echolocations within a 16 kHz to 192 kHz frequency range designed to capture calls of northeastern bat species. Each acoustic detector ran nightly for approximately 12 hours from sunset (1900 EST) to sunrise (700 EST) to capture the full extent of the foraging and travelling period.

Landscape, Local, and Event Metrics

At each sample site, I quantified local vegetation characteristics by sampling vegetation along four 25-m transects arrayed in an "+" pattern around the detector with each transect 90º from adjacent transects. I determined the first transect placement by choosing a random direction

using a compass. At 5-m intervals along each transect, I measured understory structure by counting number of woody stems in four 0.5-m height increments using a Robel pole. I measured canopy cover using a spherical densiometer and calculated percent open canopy at each sample site by averaging all 20 readings taken. I also recorded number of living trees and number of dead trees (snags) within a 2-m radius of the current sampling point along the transect. Finally, I noted tree species, diameter at breast height (DBH), total height, and status (alive or dead) of the nearest tree. I used ArcGIS to measure distance to water, distance to human structures such as houses and buildings, and distance to main roads and dirt roads from each sample point. Following each deployment event, I recorded Julian date (1 through 53 beginning on June 1st), maximum daily temperature (^{o}C), minimum daily temperature (^{o}C), average daily humidity (%), average daily wind speed (MPH), and average daily precipitation (cm).

Acoustic Data Processing

I used SonoBat version 3.2.2 software (Arcata, CA, USA) to automatically classify collected echolocation call files to species. I manually vetted all echolocation passes to confirm species identification, reduce bias, and increase identification precision (O'Farrell et al 1999). Although acoustic detectors are an important method, they are not without limitations. Misidentification rates can range from 5% to 30% and may never be eliminated among species with similar call structure, such as myotis species (Armitage and Ober 2010, Barclay 1999, Britzke et al. 2011). By manually vetting all automatic call classification, I mitigated the requirement of an estimate of software uncertainty within my analysis. I categorized bat species detection at a sample site as probable detection or no detection based on quantity and quality of call files. I classified a clear, loud echolocation call with at least 3 passes as good echolocation

calls (O'Farrell et al. 1999). Acceptable echolocation calls were lower volume and quality calls that included at least 3 passes and were identified to species by SonoBat. I categorized echolocation calls that were identified to species using SonoBat but had less than 3 passes as poor calls. At least one good or acceptable quality call for a species within a single sampling occasion indicated a probable detection (1) in that location. I classified poor quality calls or an absence of calls as not detected (0) in that site during that sampling occasion.

Modelling Co-occurrence

I applied single-species, single-season occupancy analysis to candidate models according to a set of *a priori* hypotheses using the unmarked package in R (Fiske and Chandler 2011). These models included local, landscape, and detection covariates and I ranked them using Akaike's Information Criterion. From these, I selected the single most influential covariate for each species based on ΔAIC, AICω, and beta coefficient value. I then performed conditional two-species occupancy models that incorporated the most influential covariate for each species as explanatory occupancy covariates (Richmond et al. 2010; Table 3.1). I fit 18 *a priori* models that incorporated the two covariates most important in *E. fuscus* and *M. septentrionalis* use through Program MARK version 8.1 software and the RMARK package in R (Laake 2013). I then compared these models using Akaike's Information Criterion adjusted for small sample size (AICc) and chose the highest-ranked and most parsimonious model based on ΔAICc < 2. These conditional two-species occupancy models are designed so that one species is specified as dominant over the other (subordinate) species (Richmond et al. 2010). My hypothesis was that *E. fuscus* would be the dominant species based on larger body size over the smaller *M. septentrionalis*. However, without a strong base in previous research, I chose to perform all

models twice, first with *E. fuscus* as the dominant species and then with *M. septentrionalis* as the dominant species. The species interaction factor (SIF) as defined by MacKenzie et al. (2004) provides a value to indicate independence (SIF ≈ 1), aggregation (SIF > 1), or avoidance (SIF < 1) between the two species of interest. There is currently no consensus on goodness-of-fit tests that can be used to compare model fitting for two-species occupancy models.

RESULTS

I classified 4,695 call files to species (22.7%) in 2015-2016 from 147 sample sites. From these, I detected *M. septentrionalis* on 20 sampling nights, at 15 of the 147 sites sampled. I detected *E. fuscus* on 234 sampling nights, at 109 of 147 sampling sites. Out of 588 sampling nights, *M. septentrionalis* and *E. fuscus* co-occurred on 15 nights.

Single-Species, Single-Season Occupancy Models

The most parsimonious single-species, single-season occupancy model for *E. fuscus* indicated that site use was positively associated with canopy openness (See Table 2.5). Canopy openness was negatively correlated with both trees per meter^2 (r = -0.424, P = <0.001) and average diameter at breast height (r = -0.769, P = <0.001; Figure 3.2). Therefore, canopy openness was used as a proxy for overall site clutter. The most influential and statistically supported covariate for *M. septentrionalis* site use based on the most parsimonious single-species, single-season occupancy model was sample year, which was negatively associated with use (See Table 2.4). Therefore, site clutter and sample year were chosen as exploratory covariates for following conditional two-species occupancy analysis.

Eptesicus fuscus as Dominant Species

When *E. fuscus* was specified as the dominant species, the top two models based on ΔAICc indicated that probability of site use by *E. fuscus* was associated positively with clutter and sample year (Table 3.2). Probability of site use by *M. septentrionalis* was negatively associated with sample year, both in the presence and absence of *E. fuscus.* Site use by both species together was positively associated with clutter, but 95% confidence intervals for the parameter estimate included zero. Probability of use by *M. septentrionalis* in the absence of *E. fuscus* was lower than probability of site use in the presence of *E. fuscus* ($\psi^{ME} = 0.181 \pm 0.088\ SE;\ \psi^{Me} = 0.090 \pm 0.067\ SE$; Table 3.3). The species interaction factor from the highest-ranked model shows independence of site use for both species (SIF = 1.023 $\pm$0.032 SE). The most influential covariate for model fit based on magnitude of beta coefficients was sample year as it related to probability of site use by *M. septentrionalis* in the presence of *E. fuscus* ($\beta^{\psi^{ME}} = -2.620 \pm 0.857\ SE$; Table 3.4).

Myotis septentrionalis as Dominant Species

When *M. septentrionalis* was specified as dominant, the top models indicated that probability of habitat use by *M. septentrionalis* was positively associated with site clutter and negatively associated with sample year (Appendix C.4). Probability of habitat use by *E. fuscus* in the presence of *M. septentrionalis* was also positively associated with site clutter and negatively correlated with sample year. Site use by *E. fuscus* in the absence of *M. septentrionalis* was positively associated with clutter, but not sample year. All parameter estimates except effect of

sample year on probability of use by *M. septentrionalis* has 95% confidence interval values that included zero. Probability of site use by *E. fuscus* in the absence of *M. septentrionalis* was lower than probability of site use when *M. septentrionalis* was present ($\psi^{EM} = 1.000 \pm 0.000\ SE;\ \psi^{Em} = 0.915 \pm 0.064\ SE$). Standard errors of zeros indicate poor model fitting for these models. The species interaction factor from the highest-ranked model shows independence of use by both species (SIF = 1.080 $\pm$0.064 SE). The most influential covariate for model fit based on magnitude of beta coefficients was sample year as it related to probability of site use by *E. fuscus* in the presence of *M. septentrionalis* ($\beta^{\psi^{EM}} = -902.069 \pm 6051.899\ SE$). However, the large standard error value and 95% confidence interval that includes zero also indicates poor model fitting.

DISCUSSION

E. fuscus and *M. septentrionalis* co-occur throughout the Northeast, but the extent and effects of this co-occurrence have not been examined prior to this study. Within Cape Cod National Seashore, *E. fuscus* and *M. septentrionalis* had a high level of co-occurrence with both species present on 75% of nights sampled in 2015 and 2016. Site use by *M. septentrionalis* alone was lower than probability of site use when *E. fuscus* was present. This indicates that habitat use by *M. septentrionalis* does not show a pattern of avoidance of *E. fuscus*. It also suggests that *E. fuscus* and *M. septentrionalis* partition resources successfully and feed on different insect species or groups. The species interaction factor slightly above 1 indicates that *M. septentrionalis* site use may be positively correlated with *E. fuscus* presence. This may be due to the positive association of both species to a variable that was not measured or considered in this study, such as availability of a desirable food resource. Species aggregation could also be the result of a

positive interspecific interaction, such as social information communication or habitat copying in which *M. septentrionalis* may use *E. fuscus* as an indicator of good foraging opportunities, or vice versa.

My analysis showed similar results when considering *E. fusucus* as the dominant species as when *M. septentrionalis* was specified as the dominant species and show independence of site use between *M. septentrionalis* and *E. fuscus*. These results indicate that both species are co-occurring independently and not experiencing asymmetrical competition in area or time. Poor model fitting when *M. septentrionalis* was classified as the dominant species may be due to the low number of night detections of both *M. septentrionalis* alone and in the presence of *E. fuscus*. The species interaction factor near or above 1 and higher probability of occupancy when both species are present support my conclusions that *E. fuscus* and *M. septentrionalis* are co-occurring independently.

Sample year was the most influential covariate for probability of habitat use by *M. septentrionalis* in the presence and absence of *E. fuscus* with a strong negative correlation between years. During summer of 2016, the Northeast experienced the largest outbreak of gypsy moths (*Lymantria dispar dispar*) on the Cape since 1981. Most broad-leafed forests were defoliated during peak summer months, potentially reducing insect populations in the area. The negative correlation between survey year and probability of habitat use by *M. septentrionalis* may have been due to negative effects caused by gypsy moths on insect abundance or forest cover. White-nose syndrome may also have reduced the population in this area, with effects becoming recognizable in 2016. Prior to white-nose syndrome, *M. septentrionalis* was an abundant and widespread species in the northeastern United States. Now, this species is far below population saturation which contributes to the low sample size for site detections and low

probability of detection. These issues may have caused poor model fitting when *M. septentrionalis* were specified as dominant. Without adequate goodness-of-fit tests, I was unable to assess model fitting outside of standard error and 95% confidence interval estimates. This could hinder my ability to distinguish a potential interaction, thereby limiting the scope of inference.

This research is the first to apply conditional two-species occupancy models to bats and provides an example of an approach that can be used in the future with insights into constraints and limitations. Passive acoustic monitoring can be used to successfully detect interspecific echolocations for all species present at a site and is not limited to a few target species. Bats are identified to species while actively travelling and echolocating, unlike birds that are often identified while establishing territories. This makes acoustic monitoring an ideal system for co-occurrence assessment using conditional two-species occupancy models. Future research using passive acoustic monitoring for bats should aim for a high number of night detections for both species of interest and enough nights where both species are present and detected to ensure good model fitting. This can be accomplished by increasing the number of sample sites and number of sample occasions. Mist netting in areas while simultaneously acoustically monitoring can improve detection and identification for both methods. Where species of interest are rare or at low densities, obtaining sufficient detections to ensure good model fitting may be a limitation. Wildlife managers interested in managing for bat species should consider applying conditional two-species occupancy models to previously collected acoustic identification information. In areas where species of interest aggregate or use sites independent of one another, plans may be designed that optimize habitat for multiple, co-occurring species. This baseline analysis can also reveal competitive interactions or exclusion and allow managers to focus restoration efforts on

areas where a competitor is absent. Based on these results and independent site use between *M. septentrionalis* and *E. fuscus*, Cape Cod National Seashore wildlife managers should focus on other aspects of *M. septentrionalis* recovery including establishment of high-quality diurnal maternity colony roosts and preservation of desirable foraging locations. In this area, asymmetrical competition with *E. fuscus* does not seem to be a barrier to recovery for *M. septentrionalis*.

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Table 3.A Parameter names and definitions used for conditional two-species occupancy estimation when Eptesicus fuscus is chosen as the dominant species (adapted from MacKenzie et al. 2004).

Parameter	Definition
ψ^M	Probability of site use by *Myotis septentrionalis*
ψ^E	Probability of site use by *Eptesicus fuscus*
ψ^{ME}	Probability that the site is being used by *M. septentrionalis* given that the site is occupied by *E. fuscus*
ψ^{Me}	Probability that the site is being used by *M. septentrionalis* given that *E. fuscus* is absent
p^M	Probability of detection of *M. septentrionalis* at a site given that *E. fuscus* is absent
p^E	Probability of detection of *E. fuscus* at a site given that *M. septentrionalis* is absent
r^M	Probability of detecting *M. septentrionalis* at a site given that *E. fuscus* is present
r^E	Probability of detecting *E. fuscus* at a site given that *M. septentrionalis* is present
r^{ME}	Probability of detecting *M. septentrionalis* at a site given both species are present and that *E. fuscus* was detected
r^{Me}	Probability of detecting *M. septentrionalis* at a site given both species are present and that *E. fuscus* was not detected

Table 3.B Conditional two-species occupancy model comparison following Richmond et al. (2010) for *E. fuscus* as the dominant species over *M. septentrionalis* that includes model name, Akaike's Information Criterion adjusted for small sample size (AICc), number of parameters (K), change in AICc (ΔAICc), and cumulative weight of each subsequent model. The cumulative weight for the top four models was 0.956. Covariates associated with occupancy included average canopy openness and sample year. See Table 3.1 for parameter names.

Model Name	**K**	**AICc**	**ΔAICc**	**cum. ω_i**
$\psi^{E}(canopy + year)\psi^{ME}(year)\psi^{Me}(year)$	12	864.987	0.000	0.457
$\psi^{E}(canopy)\psi^{ME}(canopy + year)\psi^{Me}(year)$	12	865.820	0.083	0.758
$\psi^{E}(canopy + year)\psi^{ME}(canopy + year)\psi^{Me}(canopy)$	13	867.220	2.233	0.908
$\psi^{E}(canopy + year)\psi^{ME}(canopy + year)\psi^{Me}(canopy + year)$	14	869.499	4.512	0.956

Table 3.C Model parameter estimates for the highest-ranked conditional two species-occupancy models based on ΔAICc, AICcω, and number of parameters for *E. fuscus* as the dominant species over *M. septentrionalis*. Includes model name, values and standard errors (SE) for probability of site use by *M. septentrionalis* given site use by *E. fuscus* (ψ^{ME}), probability of site use by *M. septentrionalis* given absence of *E. fuscus* (ψ^{Me}), and the species interaction factor (SIF).

Model Name	$\boldsymbol{\psi^{ME}}$	$\boldsymbol{\psi^{Me}}$	**SIF**
$\psi^{E}(canopy + year)\psi^{ME}(year)\psi^{Me}(year)$	0.181 (0.088)	0.090 (0.067)	1.023 (0.032)
$\psi^{E}(canopy)\psi^{ME}(canopy + year)\psi^{Me}(year)$	0.175 (0.084)	0.089 (0.066)	1.029 (0.038)
$\psi^{E}(canopy + year)\psi^{ME}(canopy + year)\psi^{Me}(canopy)$	0.181 (0.087)	0.027 (0.087)	1.042 (0.050)
$\psi^{E}(canopy + year)\psi^{ME}(canopy + year)\psi^{Me}(canopy + year)$	0.180 (0.087)	0.016 (0.060)	1.046 (0.049)

Table 3.D Model parameter and beta coefficient estimates for the top two highest-ranked conditional two species-occupancy model based on ΔAICc, AICcω, and number of parameters for *E. fuscus* as the dominant species over *M. septentrionalis*. Includes parameter name, beta coefficient estimate (β), standard error (SE), lower confidence limit, and upper confidence limit.

Model: $\psi^{E}(canopy + year)\psi^{ME}(year)\psi^{Me}(year)$				
Parameter	**β Estimate**	**SE**	**95% Confidence Interval**	
			Lower	**Upper**
ψ^{E}	-0.197	0.614	-1.400	1.006
$\psi^{E}(canopy)$	0.093	0.039	0.015	0.170
$\psi^{E}(year)$	0.585	0.637	-0.663	1.834
ψ^{ME}	0.068	0.586	-1.080	1.217
$\psi^{ME}(year)$	-2.701	0.855	-4.377	-1.024
ψ^{Me}	-2.107	1.116	-4.295	0.080
$\psi^{Me}(year)$	-0.294	0.163	-0.614	0.026
p^{E}	-0.294	0.163	-0.614	0.026
M	-0.291	1.059	-2.367	1.785
r^{E}	0.950	0.306	0.352	1.549
r^{ME}	-1.533	0.401	-2.319	-0.747
r^{Me}	-2.646	0.847	-4.306	-0.987

Model: $\psi^{E}(canopy)\psi^{ME}(canopy + year)\psi^{Me}(year)$				
Parameter	**β Estimate**	**SE**	**95% Confidence Interval**	
			Lower	**Lower**
ψ^{E}	0.241	0.391	-0.527	1.008
$\psi^{E}(canopy)$	0.082	0.033	0.016	0.147
ψ^{ME}	-0.107	0.725	-1.528	1.313
$\psi^{ME}(canopy)$	0.003	0.0128	-0.022	0.0278
$\psi^{ME}(year)$	-2.620	0.857	-4.300	-0.939
ψ^{Me}	-2.055	1.129	-4.267	0.157
$\psi^{Me}(year)$	-0.471	1.525	-3.460	2.518
p^{E}	-0.280	0.163	-0.600	0.040
p^{M}	-0.299	1.062	-2.379	1.782
r^{ME}	0.959	0.306	0.359	1.560
r^{ME}	-1.512	0.403	-2.301	-0.722
r^{Me}	-2.613	0.846	-4.272	-0.955

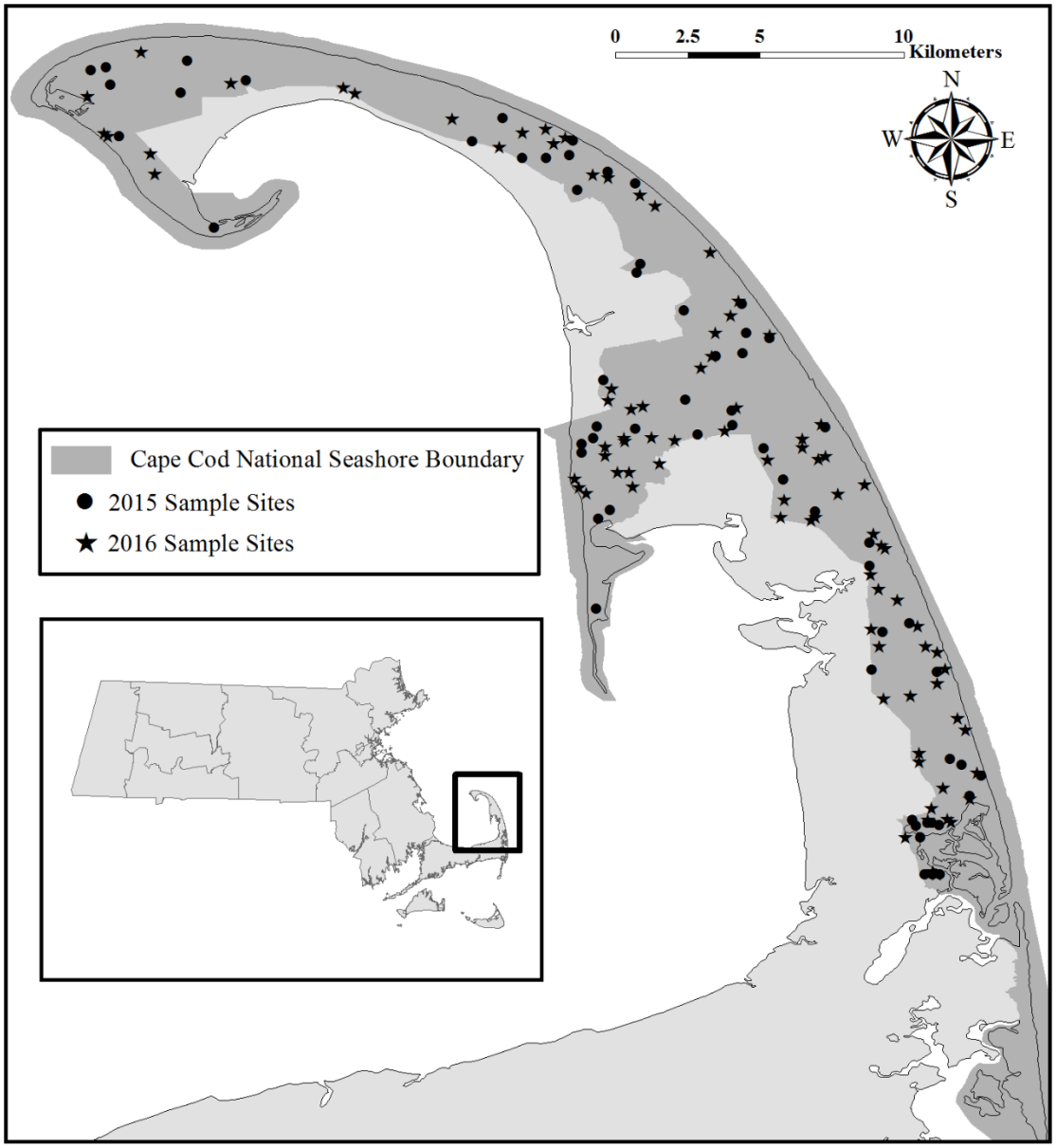

Figure 3.1 Map of study area with location of 147 passive acoustic sampling sites for summers 2015-2016 within Cape Cod National Seashore.

Figure 3.2 Pearson correlation (r) estimates comparing canopy openness against trees per meter^2 and canopy openness against average diameter at breast height for each site.

Appendix A. Initial results from passive acoustic detection deployment in 2015 and 2016

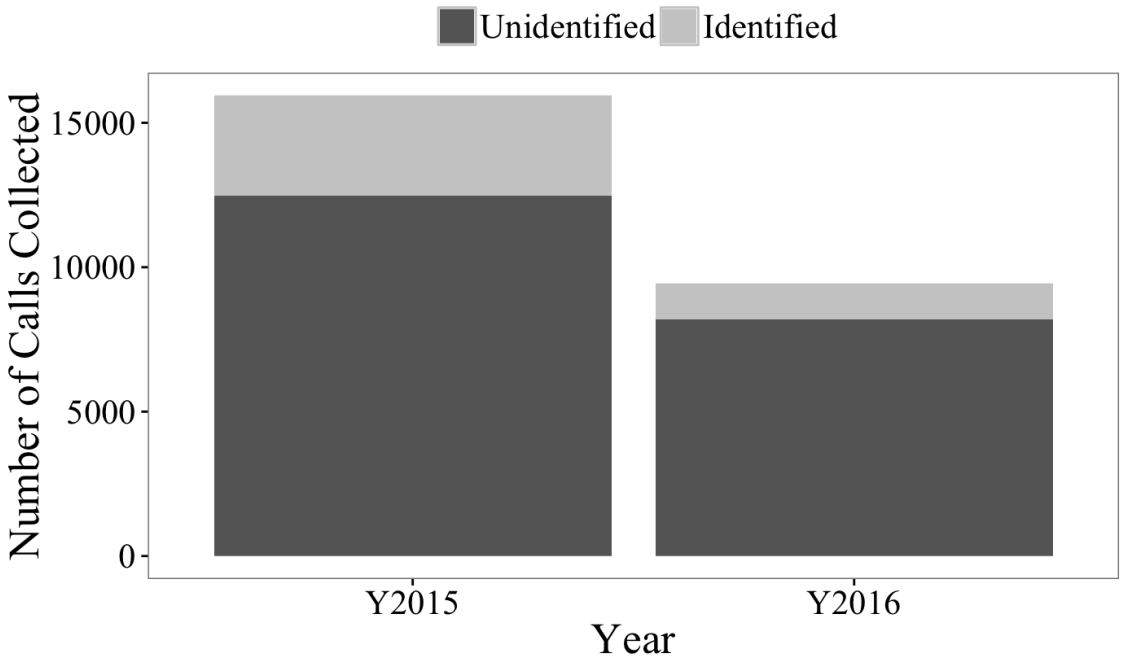

Figure A.1 Number of full-spectrum acoustic audio files collected during the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016 based on sample year and whether the audio file was identified to species.

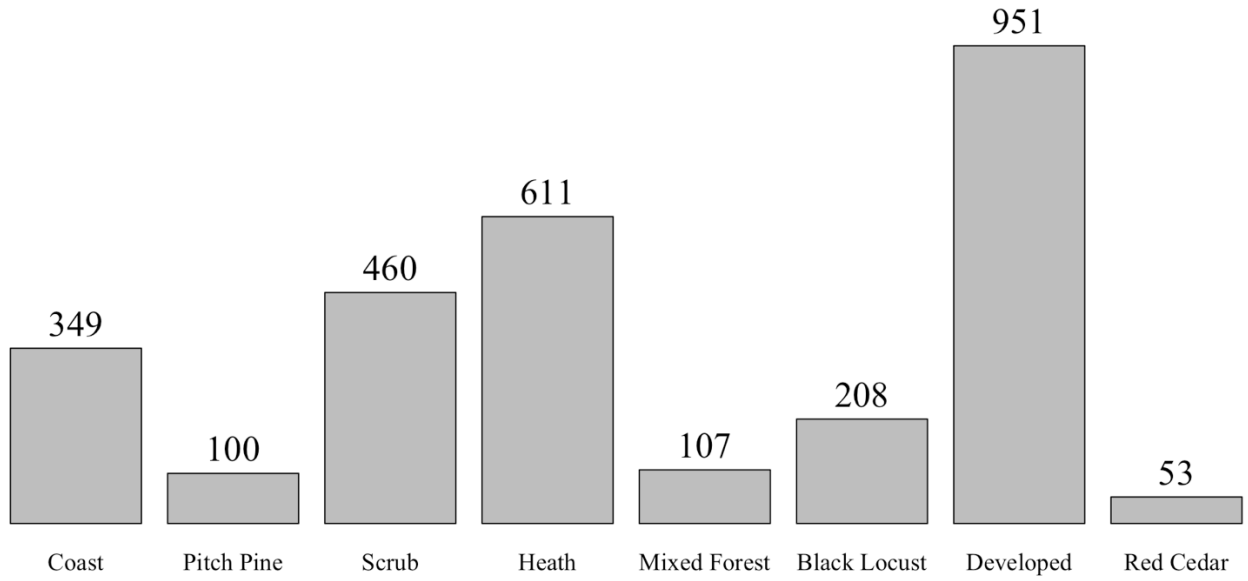

Figure A.2 Total number of full-spectrum acoustic calls classified as *Eptesicus fuscus* by vegetation category during the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

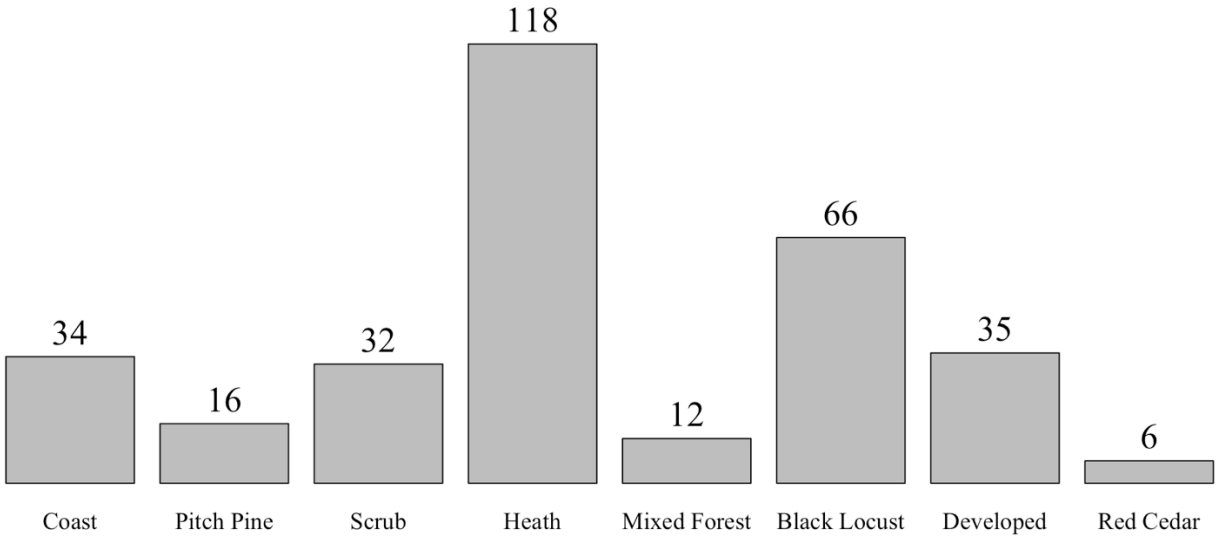

Figure A.3 Total number of full-spectrum acoustic calls classified as *Lasiurus borealis* by vegetation category during the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

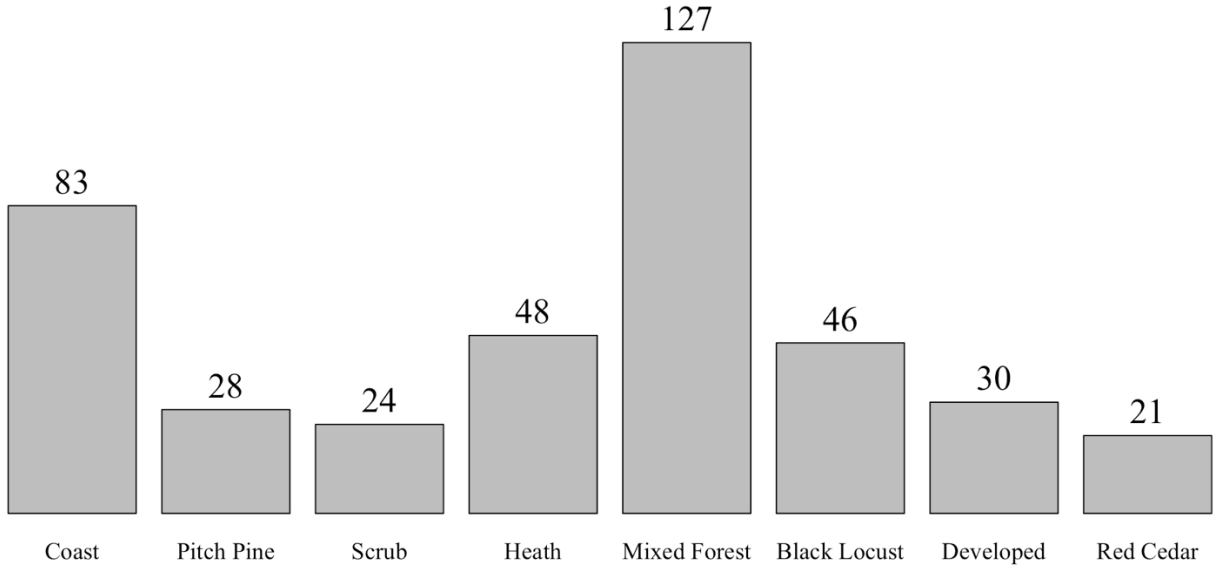

Figure A.4 Total number of full-spectrum acoustic calls classified as *Lasiurus cinereus* by vegetation category during the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

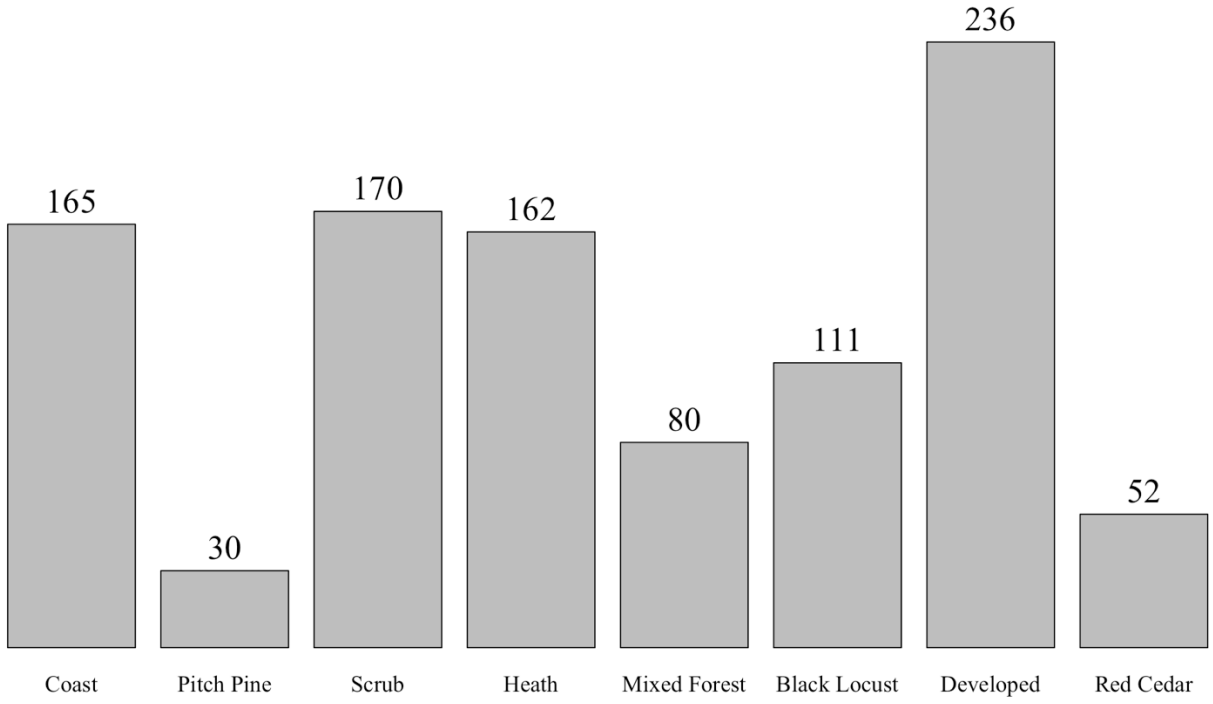

Figure A.5 Total number of full-spectrum acoustic calls classified as *Lasionycteris noctivagans* by vegetation category during the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

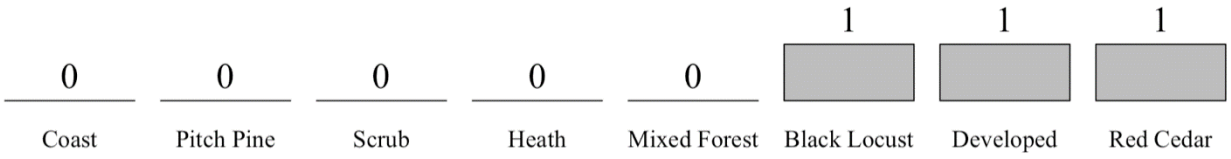

Figure A.6 Total number of full-spectrum acoustic calls classified as *Myotis leibii* by vegetation category during the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

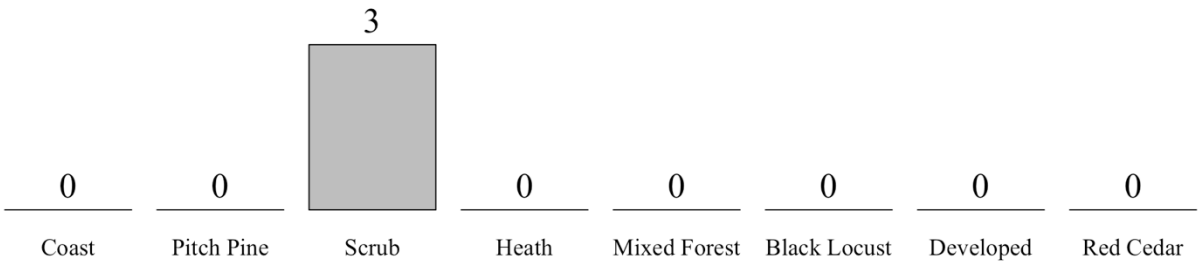

Figure A.7 Total number of full-spectrum acoustic calls classified as *Myotis lucifugus* by vegetation category during the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

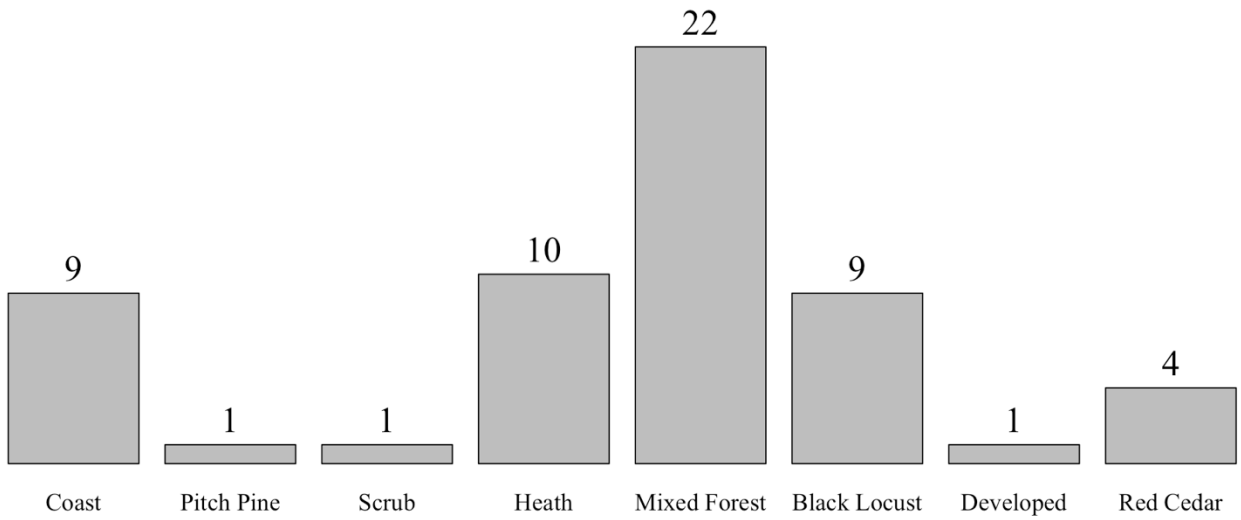

Figure A.8 Total number of full-spectrum acoustic calls classified as *Myotis septentrionalis* by vegetation category during the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

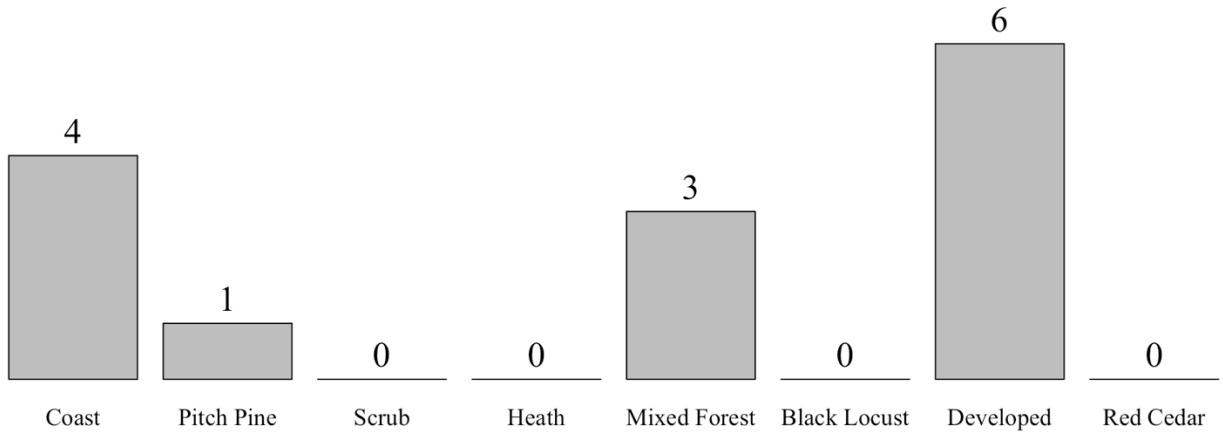

Figure A.9 Total number of full-spectrum acoustic calls classified as *Perimyotis subflavus* by vegetation category during the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

Appendix B. Species distribution maps for all species detected within Cape Cod National Seashore from 2015-2016

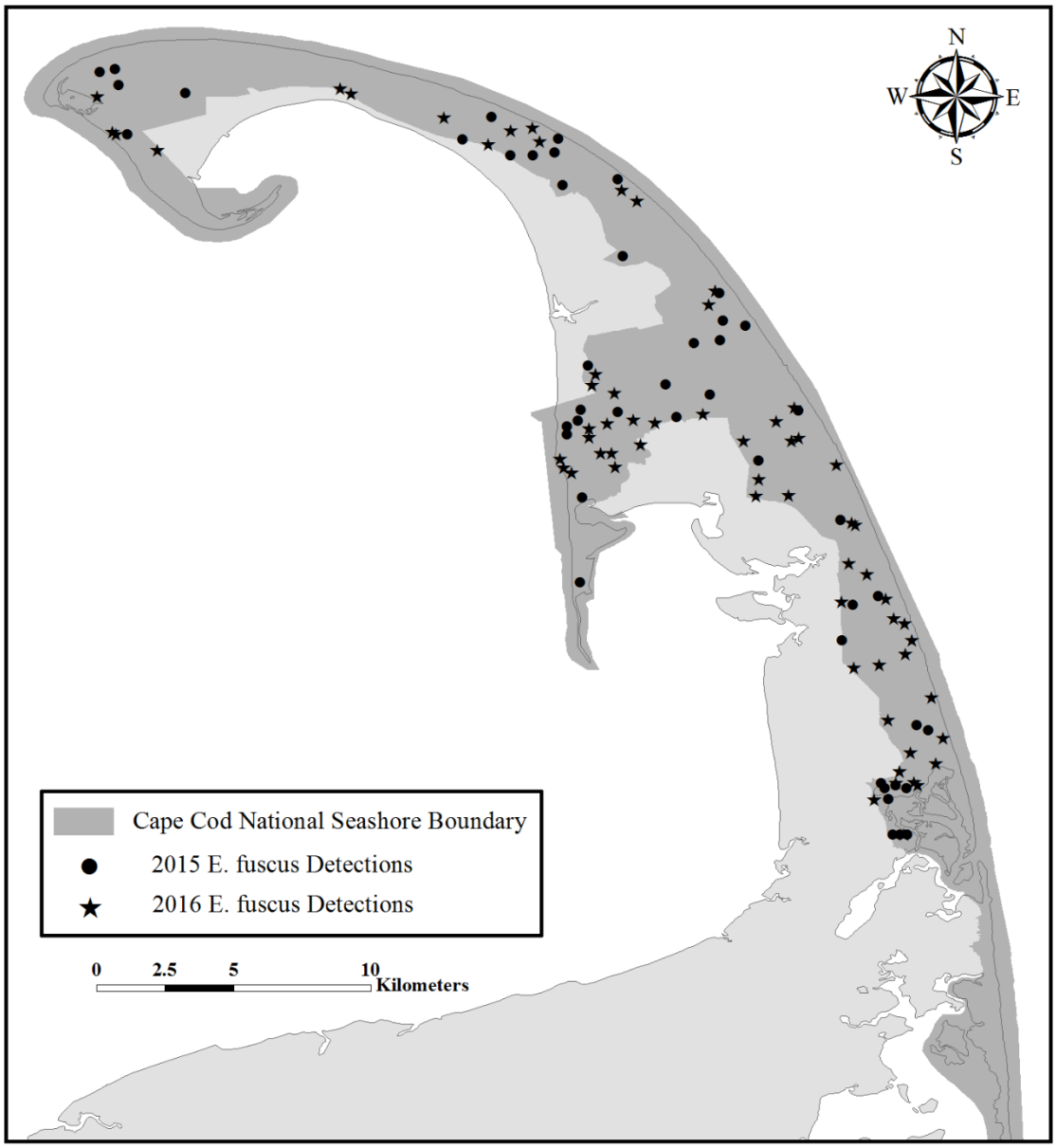

Figure B.1 Sample sites where *Eptesicus fuscus* was detected on at least one sampling night for the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

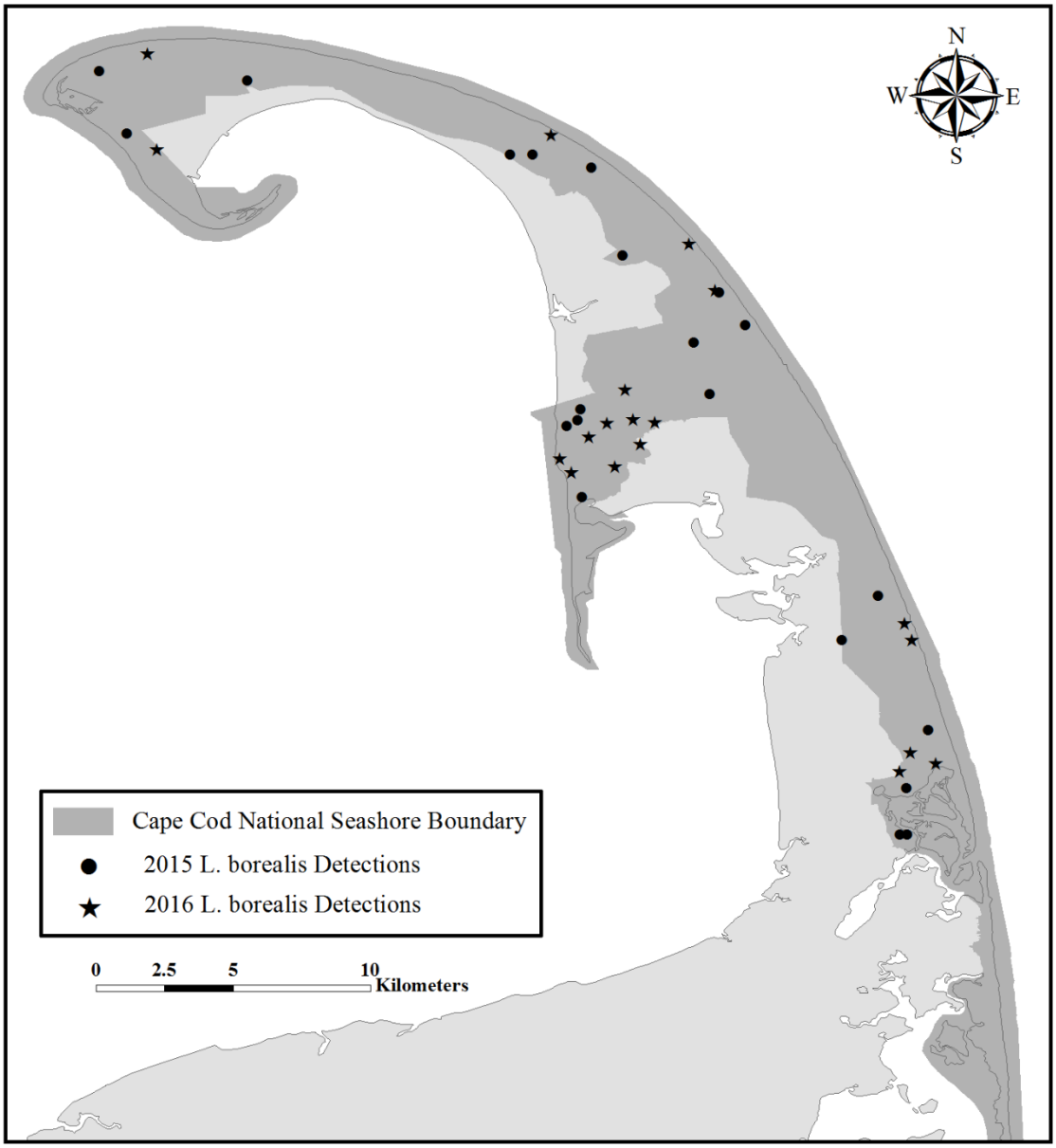

Figure B.2 Sample sites where *Lasiurus borealis* was detected on at least one sampling night for the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

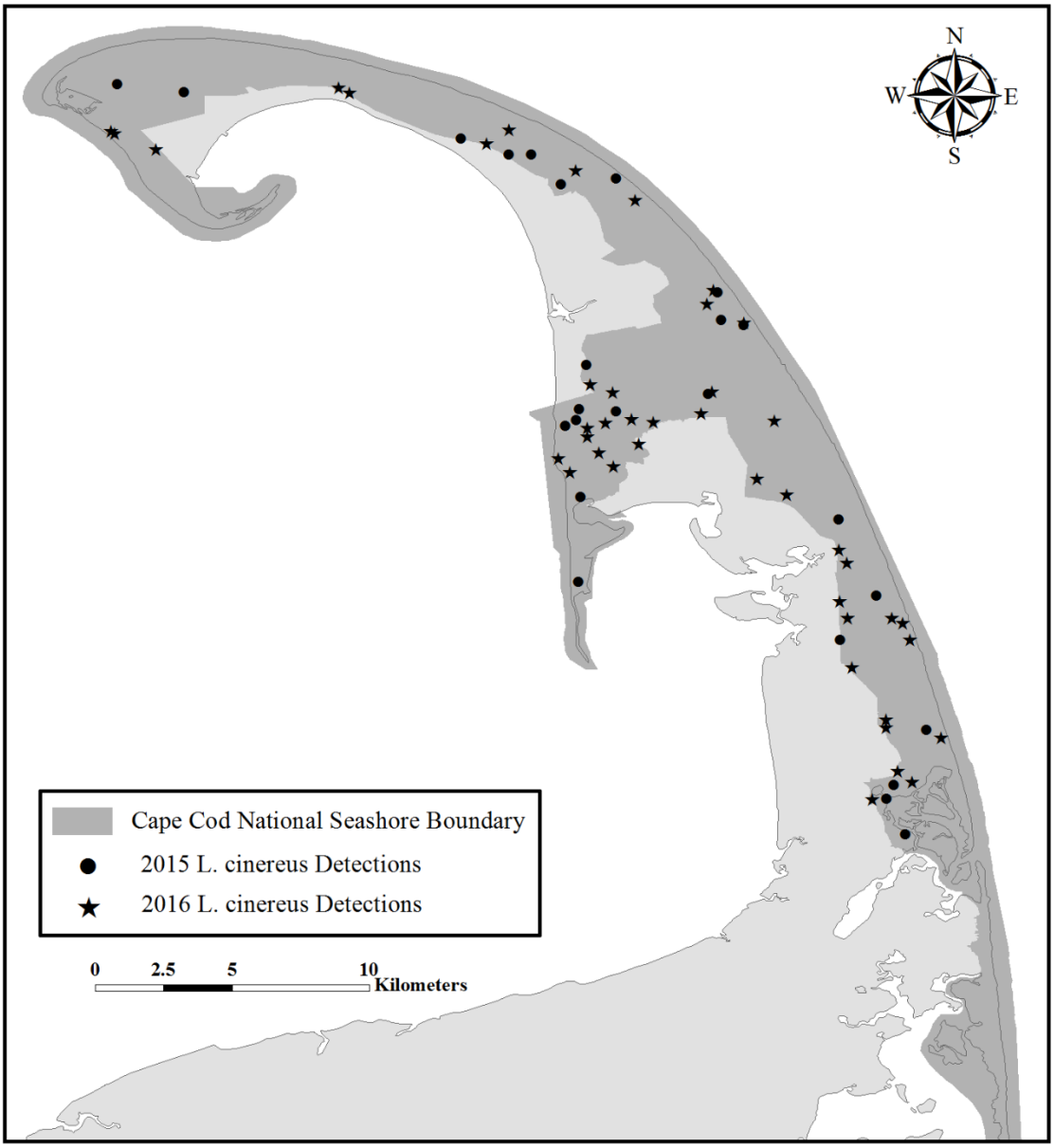

Figure B.3 Sample sites where *Lasiurus cinereus* was detected on at least one sampling night for the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

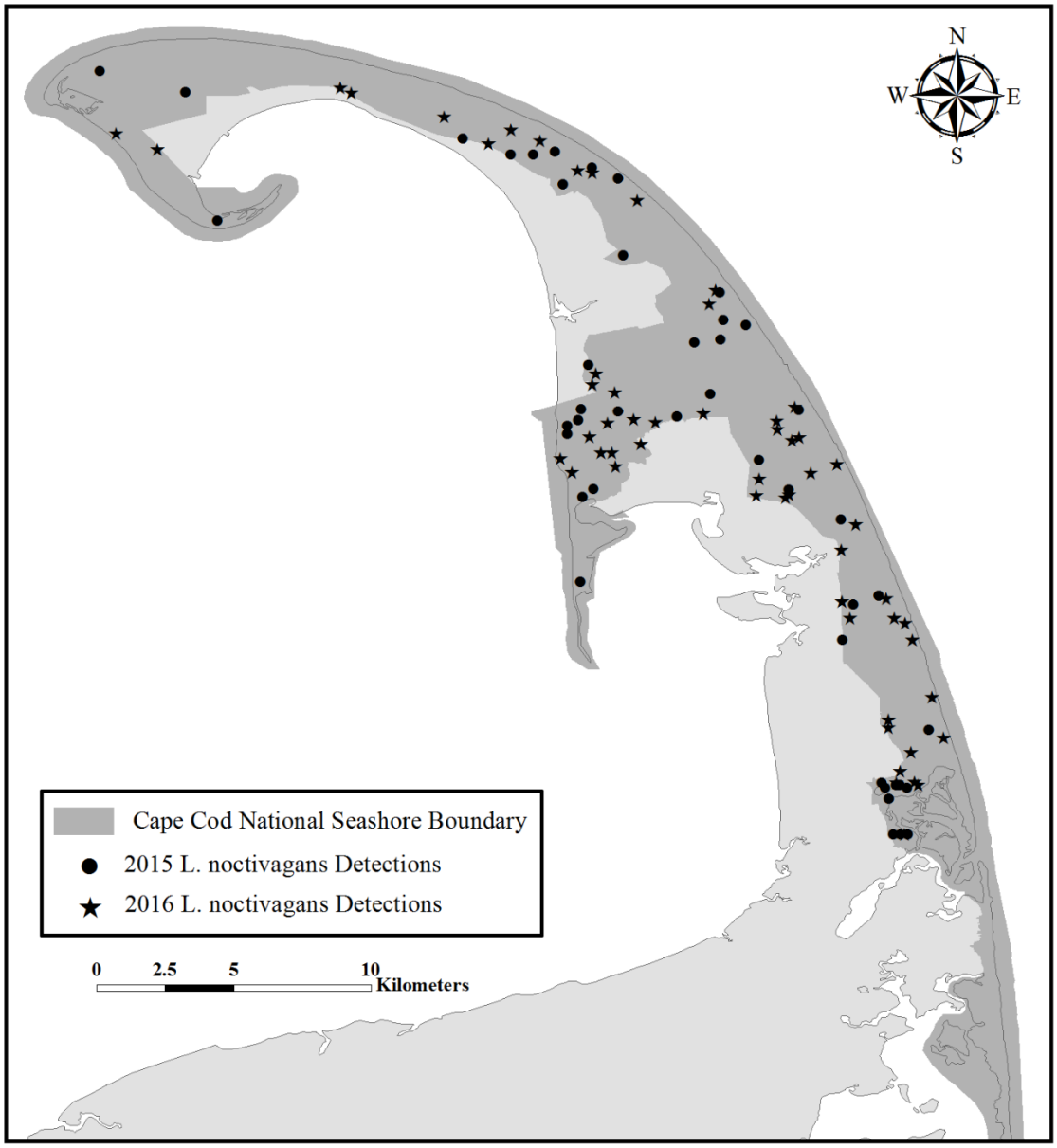

Figure B.4 Sample sites where *Lasionycteris noctivagans* was detected on at least one sampling night for the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

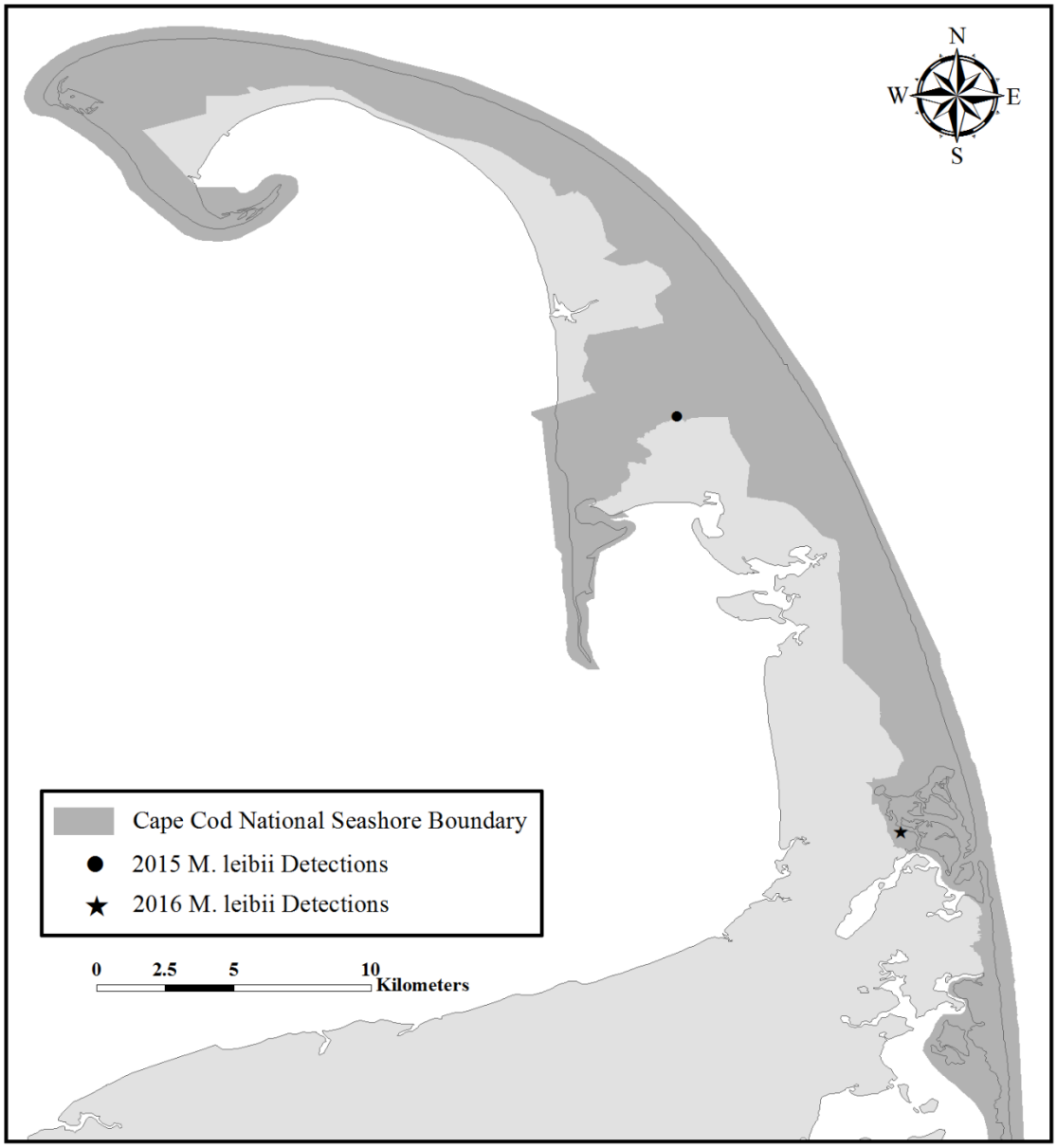

Figure B.5 Sample sites where *Myotis leibii* was detected on at least one sampling night for the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

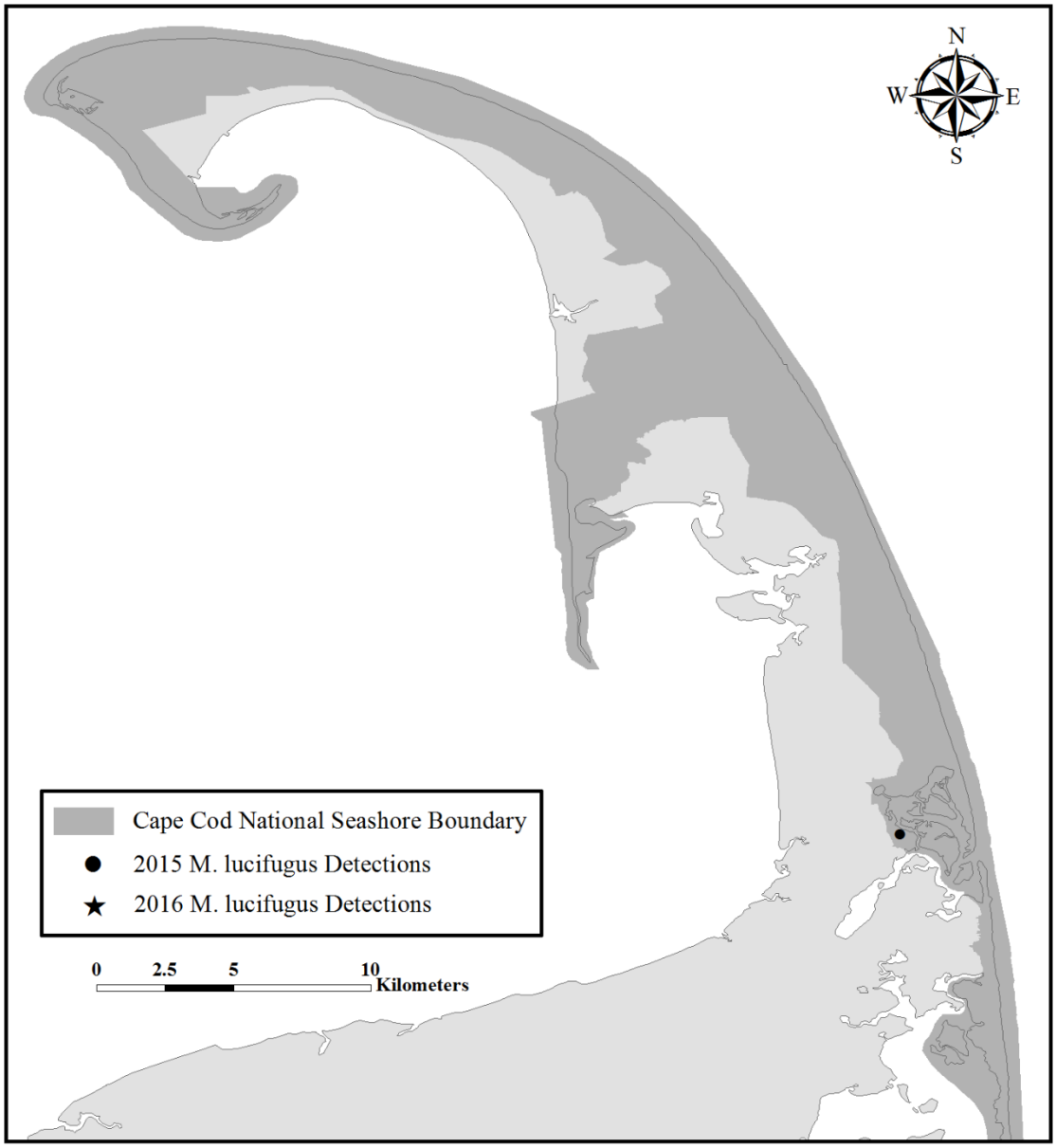

Figure B.6 Sample sites where *Myotis lucifugus* was detected on at least one sampling night for the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

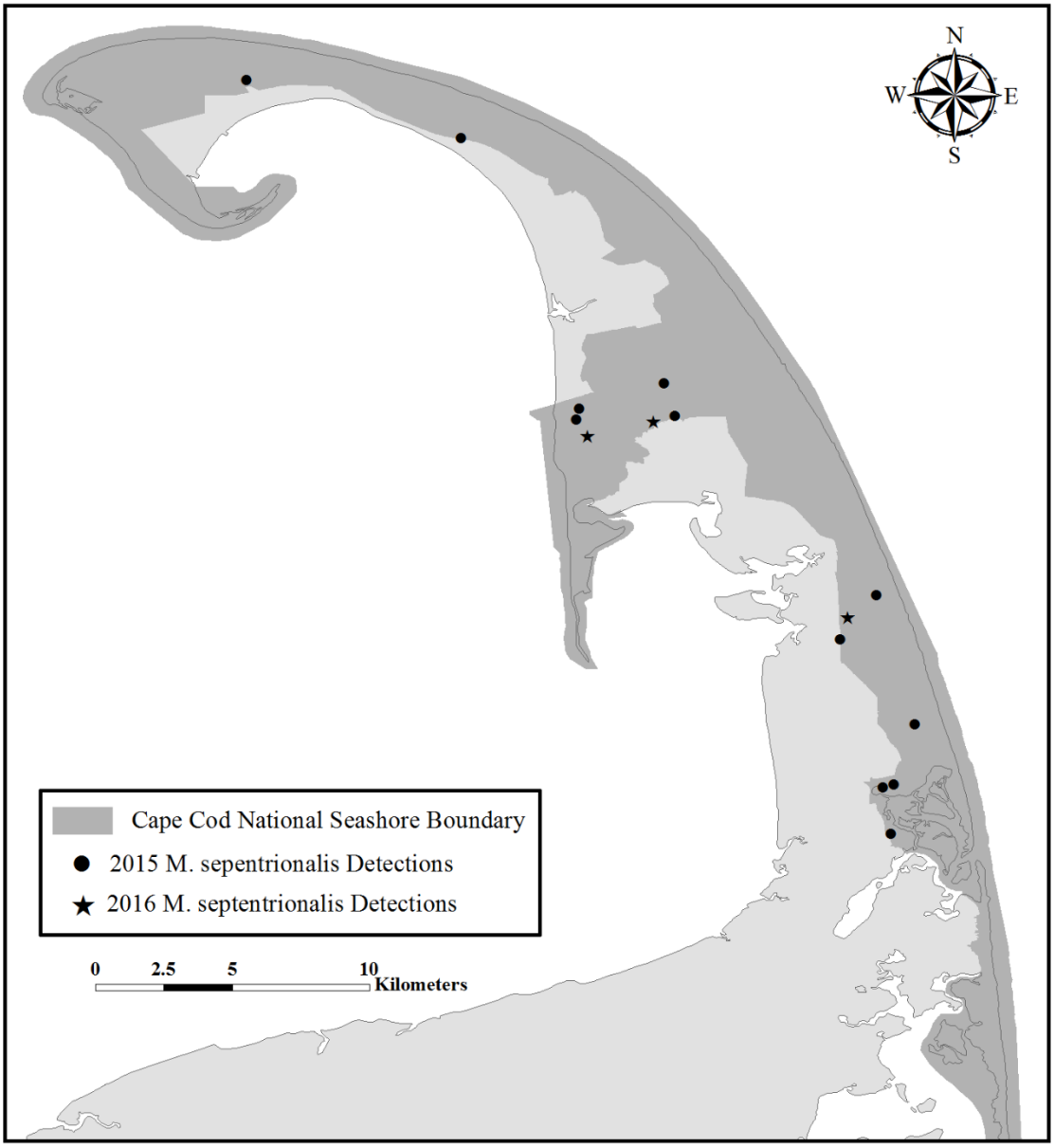

Figure B.7 Sample sites where *Myotis septentrionalis* was detected on at least one sampling night for the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

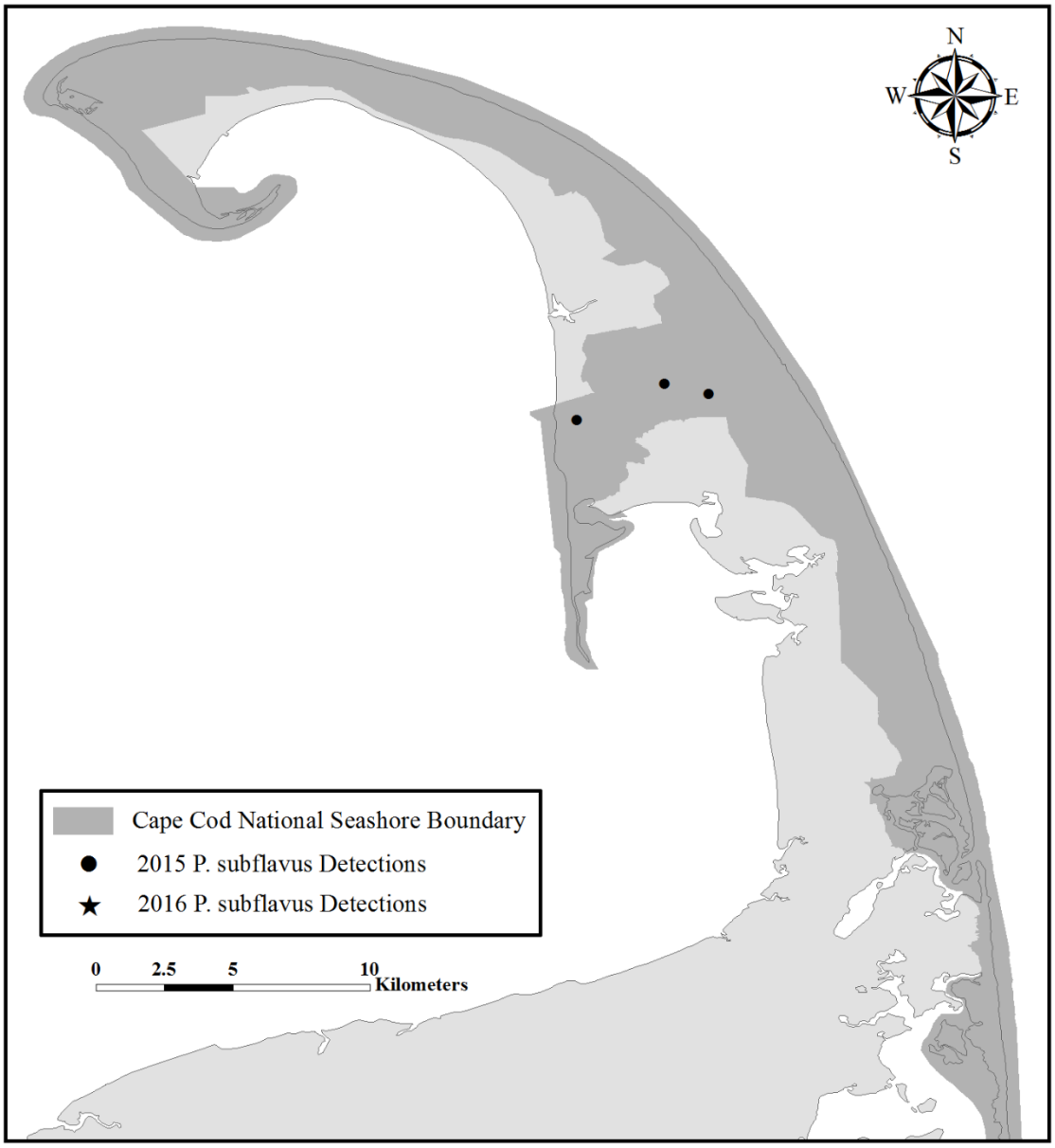

Figure B.8 Sample sites where *Perimyotis subflavus* was detected on at least one sampling night for the acoustic habitat use study conducted in Cape Cod National Seashore, Massachusetts from May-July 2015-2016.

Appendix C. Conditional two-species occupancy parameter definitions and candidate models with beta values and derived estimates

Table C.1 Parameter names and definitions used for conditional two-species occupancy estimation when *Myotis septentrionalis* is chosen as the dominant species (adapted from MacKenzie et al. 2004).

Parameter	Definition
ψ^E	Probability of site use by *Eptesicus fuscus*
ψ^M	Probability of site use by *Myotis septentrionalis*
ψ^{EM}	Probability that the site is being used by *E. fuscus* given that the site is occupied by *M. septentrionalis*
ψ^{Em}	Probability that the site is being used by *E. fuscus* given that *M. septentrionalis* is absent
p^E	Probability of detection of *E. fuscus* at a site given that *M. septentrionalis* is absent
p^M	Probability of detection of *M. septentrionalis* at a site given that *E. fuscus* is absent
r^E	Probability of detecting *E. fuscus* at a site given that *M. septentrionalis* is present
r^M	Probability of detecting *M. septentrionalis* at a site given that *E. fuscus* is present
r^{EM}	Probability of detecting *E. fuscus* at a site given both species are present and that *M. septentrionalis* was detected
r^{Em}	Probability of detecting *E. fuscus* at a site given both species are present and that *M. septentrionalis* was not detected

Table C.2 Conditional two-species occupancy model comparison following Richmond et al. (2010) for *M. septentrionalis* as the dominant species over *E. fuscus* that includes model name, Akaike's Information Criterion adjusted for small sample size (AICc), number of parameters (K), change in AICc (ΔAICc), and cumulative weight of each subsequent model The cumulative weight for the top two models was 0.969. Covariates associated with occupancy included average canopy openness as a proxy for site clutter and sample year. See Table 3.1 for parameter names.

Model Name	**K**	**AICc**	**ΔAICc**	**cum. ω_i**
$\psi^{M}(clutter + year)\psi^{EM}(clutter + year)\psi^{Em}(clutter)$	12	852.116	0.000	0.567
$\psi^{M}(clutter + year)\psi^{EM}(clutter + year)\psi^{Em}(clutter + year)$	13	852.805	0.689	0.969

Table C.3 Model parameter estimates for the highest-ranked conditional two species-occupancy models based on ΔAICc, AICcω, and number of parameters for *M. septentrionalis* as the dominant species over *E. fuscus*. Includes model name, values and standard errors (SE) for probability of site use by *E. fuscus* given site use by *M. septentrionalis* (ψ^{EM}), probability of site use by *E. fuscus* given absence of *M. septentrionalis* (ψ^{Em}), and the species interaction factor (SIF).

Model Name	$\boldsymbol{\psi^{ME}}$	$\boldsymbol{\psi^{Me}}$	**SIF**
$\psi^{M}(clutter + year)\psi^{EM}(clutter + year)\psi^{Em}(clutter)$	1.000 (0.000)	0.915 (0.064)	1.080 (0.064)
$\psi^{M}(clutter + year)\psi^{EM}(clutter + year)\psi^{Em}(clutter + year)$	1.000 (0.000)	0.930 (0.066)	1.064 (0.064)

Table C.4 Model parameter and beta coefficient estimates for the top two highest-ranked conditional two species-occupancy model based on ΔAICc, AICcω, and number of parameters for *M. septentrionalis* as the dominant species over *E. fuscus*. Includes parameter name, beta coefficient estimate (β), standard error (SE), lower confidence limit, and upper confidence limit.

Model: $\psi^{M}(canopy + year)\psi^{EM}(canopy + year)\psi^{Em}(canopy)$				
Parameter	**β Estimate**	**SE**	**95% Confidence Interval**	
			Lower	**Upper**
ψ^{M}	-0.823	0.487	-1.777	0.132
$\psi^{M}(clutter)$	0.009	0.011	-0.013	0.031
$\psi^{M}(year)$	-2.258	0.693	-3.616	-0.901
ψ^{EM}	-332.271	2084.334	-4417.566	3753.0238
$\psi^{EM}(clutter)$	103.322	5.297	92.941	113.704
$\psi^{EM}(year)$	-902.069	6051.899	-12763.792	10959.653
ψ^{Em}	0.353	0.415	-0.461	1.166
$\psi^{Em}(clutter)$	0.064	0.032	0.002	0.127
p^{M}	-0.306	1.021	-2.307	1.694
p^{E}	-0.248	0.153	-0.547	0.052
r^{M}	-1.555	0.378	-2.296	-0.815
r^{EM}	2.015	0.753	0.539	3.490
r^{Em}	1.003	0.344	0.328	1.678

Model: $\psi^{M}(canopy + year)\psi^{EM}(canopy + year)\psi^{Em}(canopy + year)$				
Parameter	**β Estimate**	**SE**	**95% Confidence Interval**	
			Lower	**Upper**
ψ^{M}	-0.741	0.503	-1.727	0.244
$\psi^{M}(clutter)$	0.009	0.011	-0.013	0.031
$\psi^{M}(year)$	-2.323	0.702	-3.700	-0.947
ψ^{EM}	-372.705	2712.175	-5688.568	4943.158
$\psi^{EM}(clutter)$	111.487	0.949	109.627	113.347
$\psi^{EM}(year)$	-1054.669	0.144	-1054.950	-1054.387
ψ^{Em}	-0.398	0.738	-1.844	1.048
$\psi^{Em}(clutter)$	0.078	0.041	-0.002	0.158
$\psi^{Em}(year)$	0.926	0.740	-0.524	2.376
p^{M}	-0.311	1.031	-2.332	1.710
p^{E}	-0.259	0.152	-0.556	0.038

r^{M}	-1.607	0.377	-2.346	-0.868
r^{EM}	2.015	0.753	0.539	3.490
r^{Em}	0.971	0.343	0.300	1.643

CURRICULUM VITAE

Jessica Fletcher

(248) 229-4739 ✦ jessiefletch79@gmail.com

EDUCATION

State University of New York College of Environmental Science & Forestry Syracuse, New York 13210 — Expected December 2017

Masters of Science Candidate in Fish and Wildlife Biology and Management

Cumulative 3.936 out of 4.000

University of Michigan
Ann Arbor, Michigan 48109 — Graduated May 2014

Bachelor of Science Dual Degree in Ecology and Evolutionary Biology and Environmental Sciences

Minor: Music

Cumulative 3.252 out of 4.000

Ernest W. Seaholm High School
Birmingham, Michigan 48009 — Graduated May 2010

PROFESSIONAL AND RESEARCH EXPERIENCE

Agassiz National Wildlife Refuge
22996 290th St NE, Middle River, MN 56737 — May 2017 – August 2017

Directorate Fellow, 40 hours/week

- Collected comprehensive bathymetric information within three management units using Trimble Global Navigation Satellite Systems equipment
- Created contour maps, water-depth inundation tables, depth proportion figures, and stage volume figures for three management units
- Assisted Minnesota DNR with Canada goose (*Branta canadensis*) banding through paneling, capturing, and transporting individuals to be banded
- Compiled all bathymetric data and collection protocols into comprehensive reports for future use at Agassiz National Wildlife Refuge
- Presented results and potential future directions to refuge staff and interested stakeholders

State University of New York College of Environmental Science & Forestry
1 Forestry Drive, Syracuse, New York 13210 — September 2015 – December 2017

Masters of Science Candidate and Teaching Assistant, 40 hours/week

- Analyzed acoustic detection/non-detection data in an occupancy framework to determine habitat use with a focus on bat species co-occurrence
- Final analysis assisted the National Park Service at Cape Cod National Seashore in their development of management strategies for all bats, including the federally threatened northern long-eared bat (*Myotis septentrionalis*)
- Primary teaching assistant for the undergraduate ornithology course responsible for over 60 undergraduate students and four undergraduate teaching assistants

Cape Cod National Seashore June 2015 – August 2015
99 Marconi Site Road, Wellfleet, Massachusetts 02667 June 2016 – August 2016
Graduate Student Researcher and Crew Leader, 40 hours/week

- Managed day-to-day bat acoustic detection deployment and created daily schedules for two undergraduate research technicians
- Independent set-up and tear-down of mist nets over 10 net nights. Safe removal and processing of 28 bats including experience with the federally threatened northern long-eared bat (*Myotis septentrionalis*)
- Deployed Wildlife Acoustics SM3 bat detectors at 147 randomly generated, unique locations along the Cape Cod National Seashore
- Used radio telemetry equipment to tag and track *Myotis septentrionalis* individuals to day roosts

Organization for Bat Conservation September 2014 – January 2015
Volunteer Animal Keeper, ~5 hours/week

- Responsible for preparing and distributing food to resident animals and maintenance of all enclosures
- Trained in proper handling of bats and other small mammals for routine medical checks and some hand-feeding
- Authorized for bat handling following completion of the pre-exposure rabies vaccine in October 2014

Big Oaks National Wildlife Refuge
1661 W Jpg Niblo Rd, Madison, Indiana 47250 May 2014 – August 2014
Volunteer Biological Intern, 40 hours/week

- Conducted extensive invasive plant species identification and inventory as part of an invasive species removal project
- Used ArcGIS to pinpoint locations of interest and then conducted an invasive plant species identification and inventory project that spanned over 3,000 acres of forest at Big Oaks NWR
- Surveyed bat populations on the refuge using the AnaBat acoustical monitoring system in part to monitor the success of the federally endangered Indiana bat (*Myotis sodalis*)
- Participated in bird surveys as part of an occupancy monitoring program for the near threatened Henslow's sparrow (*Ammodramus henslowii*)
- Assisted in demographic studies during a recovery program of the state-endangered crawfish frog (*Lithobates areolatus*) that included construction of a drift fence, collection of recently metamorphosed tadpoles, and recording of quantitative measurements

Tibbetts Laboratory, Ann Arbor, Michigan September 2012 – April 2014
Undergraduate Researcher, 9 hours/week

- Participated in video analysis of experiments that sought to understand the presence or absence of individual recognition in *Polistes fuscatus* worker wasps
- Collected and converted qualitative data into usable, quantitative data that was entered into MS Excel
- Assisted in building wooden animal enclosures using power tools

Teaching and Inspiring Environmental Stewardship January 2014 – April 2014
440 Church Street, Ann Arbor, Michigan 48109
Docent, ~5 hours/week

- Led tours of the Gold LEED certified Samuel Trask Dana building that highlighted environmentally sustainable advances
- Provided groups with suggestions for environmentally friendly changes in their homes and schools
- Tailored interpretative activities to different developmental levels including middle school, high school, and undergraduate participants

CONFERENCES

2017 Annual Meeting of the Northeast Bat Working Group	Amherst, Massachusetts January 2017
46th Annual North American Symposium on Bat Research	San Antonio, Texas October 2016
American Society of Mammalogists 96th Annual Meeting	Minneapolis, Minnesota June 2016
2016 Annual Meeting of the Northeast Bat Working Group	Baltimore, Maryland January 2016
The Wildlife Society 22nd Annual Conference	Winnipeg, Manitoba, Canada October 2015

PRESENTATIONS

Fletcher, J.J., and Farrell, S.L. October 2016. Habitat use and species composition of bats in a northeastern coastal plain ecosystem. Paper presented at the 46th annual North American Symposium on Bat Research, San Antonio, TX.

Fletcher, J.J., and Farrell, S.L. June 2016. Habitat use and species composition of bats in a northeastern coastal plain ecosystem. Poster presented at the American Society of Mammalogists 96th annual meeting, Minneapolis, MN.

Fletcher, J.J., and Farrell, S.L. January 2016. Characterizing habitat use and species composition of bats in Cape Cod National Seashore. Paper presented at the 2016 annual meeting of the Northeast Bat Working Group, Baltimore, MD.

Fletcher, J.J., and Farrell, S.L. October 2016. The bats of Cape Cod. Paper presented at the Science in the Seashore Symposium, Eastham, MA.

Fletcher, J.J. and Walker, B.A. July 2014. Acoustic monitoring of bats in Big Oaks NWR. Paper presented at the U.S. Fish & Wildlife Service Intern Workshop, Minneapolis, MN.

AWARDS

ESF Graduate Student Travel Grant, October 2016

Edna Bailey Sussman Graduate Fellowship, April 2016

Fink Career Fellowship, February 2016

SUNY ESF Graduate Student Association Travel Grant, December 2015

TRAINING AND CERTIFICATION

Project Wet, Project Wild, and Project Learning Tree Teaching Certification, February 2014

Autonomous Aerial Acoustic Recording Systems (AAARS) Bird Monitoring Training, June 2014

S-130: Firefighter Training and S-190: Introduction to Wildland Fire Behavior Course Completion, August 2014

SonoBat version 3.2.2 Training, November 2015

SKILLS

- Experienced conducting data analysis using MS Excel, MS Access, ArcGIS, R, Sonobat, Mark, and Presence
- Comfortable and confident with data collection, entry, and analysis
- Effective, enthusiastic, and experienced public speaker